The Impacts of Edge Effects and Other Disturbances on Tasmanian Coastal Heathlands

by

Mona Loofs-Samorzewski BSc (Hons) ANU

Submitted in fulfilment of the requirements for

the degree of Doctor of Philosophy

School of Geography and Environmental Studies

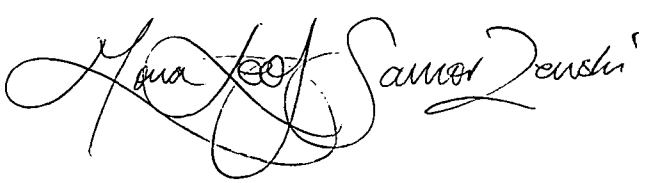
University of Tasmania

July 2003

Declaration

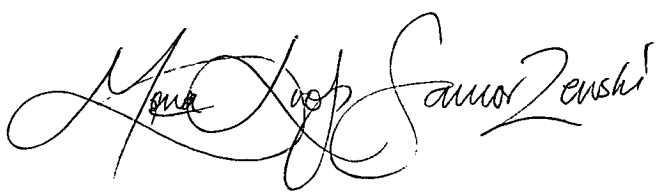
This thesis contains no material that has been submitted for any other award or degree at any other institution.

To the best of my knowledge and belief, this thesis does not contain any material published or written by any other person except where due reference is made in the text.



Authority of Access

This thesis may be made available for loan and limited copying in accordance with the *Copyright Act 1968*.



Abstract

Widespread clearing of Tasmanian heathland for pasture and crops has resulted in small remnants within the agricultural landscape. The conventional wisdom has been that such small remnants are susceptible to adverse edge effects, particularly those related to nutrient accession, and thus are not worth conserving. Previous research into the addition of nutrients in heathlands has involved large amounts of fertiliser. In this thesis realistic experimental manipulation in the field and widespread observation of boundaries are used to determine whether there could be a future for small heath remnants.

The effects of irrigating a coastal heathy woodland with sewage water were monitored. The addition of small amounts of nutrients in the wastewater caused a small increase in live plant cover but no major plant death or changes in species composition. In a manipulative field experiment, small amounts of fertilisers were added to simulate nutrient drift from pasture topdressing, and were combined with fire and weed seed treatments. There were no effects on cover, species richness and species composition. In a second manipulative field experiment, fire-fighting foam was applied in combination with burning. Foam effects included increases and decreases in canopy growth of different species, a reduction in species richness, reduced flowering and leaf damage. In all three experiments, exotic species either did not invade over the period of the study or, when weed seeds were sown, failed to establish. In a survey, the observations of heath–pasture boundaries showed that different management regimes affected the condition of coastal heath remnants. The major disturbance that encouraged weed invasion in heathlands was an increase in soil fertility. Stock grazing, native herbivore grazing and physical disturbance also adversely affected heath condition. However, edge effects in the form of increased nutrients and the presence of exotic species did not penetrate more than a few metres into heathland. In summary, with careful management, small heathland remnants can be quite resistant to weed invasion and are worth conserving.

Acknowledgments

I would like to thank Jamie Kirkpatrick for his excellent supervision, encouragement, expertise and inspiration. Support was provided by the School of Geography and Environmental Studies. In particular, Denis Charlesworth for soil analysis; Kate Charlesworth for Resource Centre assistance; David Sommerville, Lee Reid, Kate Murphy, Helen Reid and Moya Kilpatrick for administrative support. Mike Harlow and Darren Turner assisted with all computer matters. Vehicles were supplied by Geography and University Car Pools.

The heathland irrigation at Stieglitz was brought to my attention by David Dettrick. The Break O'Day Council provided fieldwork equipment and John Brown gave permission to access the Stieglitz Sewage Ponds. Richard Mason explained the sewage irrigation and toured the site with me. Trevor Berryman and other Council employees also provided assistance. I am grateful for the accommodation given by Polly and Mary Buckhorn and the MacDonald family. Louise Gilfedder read and commented on a chapter draft.

John Shelverton of the Housing Department gave me permission to work at Huntingfield. Thanks also to the Huntingfield/Coffee Creek Landcare Group for information and advice about the area.

The Parks and Wildlife Service Tasmania (Department of Primary Industries, Water and Environment) assisted with transport, fire-fighting equipment, fire-fighting foam and accommodation. For the burns at Huntingfield and Mt William, thanks to Mark Whitney and the Fire Crew. For helping organise the burns, fieldwork assistance and comments on one chapter thanks to Adrian Pyrke. Rod Pearse loaned information on the history of management of Mt William National Park and David Keith provided assistance with experimental design. Robyn Adams read and commented on an early chapter draft.

Travel funds for the heath–pasture boundary survey were provided by LWRRDC. The Parks and Wildlife Service Tasmania provided access to aerial photos of Flinders Island. Tim Rudman gave me advice and information about *Phytophthora cinnamomi*. Many thanks to all the following for allowing me onto their properties and answering the management questionnaires: Malcolm Aitken, Mr Burrows, Rob Cook, Ron Finch, Rodney Fowler, Gilbert Green, Graham Hill, Phil and Ken Irby, John McNab, Max Medwin, Dave Montgomery, David and Jill Murray, Rodney Pitchford, Roy and Dolly Peddle, Derek Smith, and Alan Stackhouse. Special thanks to Kevin and Helen Haines, Stuart Dudgeon and Eric Nicholls. Also, I very much appreciate Mandy Larby, Steve Cronin and Tim Gunn for being in the right place at the right time. Much appreciation also goes to Sophie Underwood for accommodation on Flinders Island and help with fieldwork practicalities.

Many other people assisted me in many ways during the course of my PhD. Phil Barker helped me find direction and define my project in the early days. Hans Wapstra was invaluable in identifying orchids. Dennis Morris and Alex Buchanan at the Herbarium identified endless streams of scrappy little plant samples. Jonathon Basil Marsden-Smedley provided weather information and fire measurements during the Huntingfield burn. David Ratkowsky gave me much needed advice about statistics. Anita Wild assisted with ArcView and Stephen Bresnehan created the maps. Kathy Allen, Matt Appleby, Kerry Bridle, Michael MacDonald, Sapphire McMullan-Fisher and Pep Turner read and commented on chapters and provided excellent feedback and discussion. Thanks also to Emma Pharo, Lily Sekuljica, Janet Smith and Johnny Tasirin. My parents Sigi and Helmut provided unconditional financial and moral support that was invaluable.

Finally, this project would not have been possible without the world's best field assistant, meika von samorzewski. Thankyou for doing most of the driving, getting us unbogged, finding interesting routes to drive home on, taking soil samples, slashing fire breaks, writing notes, holding the umbrella in the rain, always being able to find a quadrat if I'd lost it, making dinner, taking photos, writing about Cobblers Rocks, pepping me up when I needed it, hassling me when I needed it, letting me cry on his shoulder when I needed it, just for being there, always, with his perfect company, and without whom this PhD could not have been done, of course.

Table of contents

List of figures

List of tables

1. Introduction

WHAT IS HEATHLAND?

Definitions and distributions

Heathland is defined as dense to mid-dense vegetation dominated by evergreen scleromorphic shrubs less than two metres high (Specht 1981b; Kirkpatrick and Harris 1999b). The climatic range of heathlands is one of the widest of any vegetation community in the world, ranging from coastal areas to alpine regions and from the tropics to the subarctic. The single common environmental factor controlling the distribution of heathlands is the infertility of the soils on which they are found (Specht and Rayson 1957; Grundon 1972; Groves 1981a; Specht 1981b). The 'heath' families of Diapensiaceae, Empetraceae, Ericaceae, Grubbiaceae, Prionotaceae and Vaccinaceae are prevalent in the Northern Hemisphere and can dominate the community. In Australia, the prominent shrub families are Epacridaceae, Myrtaceae, Proteaceae and Rhamnaceae (Specht 1981b). Heath also commonly forms an understorey, particularly beneath scleromorphic small trees or tall shrubs.

Many heathlands are found in areas that have a climate similar to the Mediterranean Basin in Europe, that is, a cool, wet winter alternating with a hot, dry summer (di Castri 1981; Specht and Moll 1983). Mediterranean-type ecosystems are found in four places other than Europe, namely California, central Chile, southern Africa and southwest and southern Australia, all at latitudes of 30–40° (di Castri 1991). The vegetation is dominated by woody shrubs with evergreen scleromorphic leaves, with the occasional overstorey of small trees and an understorey of annuals and herbaceous or woody perennials (Specht and Moll 1983; di Castri 1991). Generally, the vegetation formation in these regions is called Mediterranean shrubland, having a grassy understorey and being found on relatively fertile soils. Shrublands are variously called 'maquis' or 'garrigue' in Europe, 'matorral' in Chile, 'renosterveld' in South Africa, and 'mallee' in southern Australia (Specht and Moll 1983; di Castri 1991). However, on highly leached (oligotrophic) soils a heath understorey or true heath develops (Specht and Moll 1983). This type of soil is present in South Africa, to some extent in California, in the Mediterranean Basin and in southern and southwestern Australia. The vegetation growing on oligotrophic soil is called 'fynbos' in the drier regions of South Africa, 'chaparral' in California, 'heathland' in eastern Australia and Europe, 'mallee heathland' or 'mallee-broombush' in southern Australia and 'kwongan' in the Australian southwest (Specht and Moll 1983; di Castri 1991).

In Australia, heathlands are also found from tropical to temperate climates, from subhumid to humid regions and from coastal to alpine habitats (Groves and Specht 1965; Specht 1981a). Alpine and sub-alpine heathlands are present in Tasmania, Victoria and New South Wales (Costin *et al.* 1982; Kirkpatrick and Harris 1999b). Large areas of lowland heath are found in the Mediterranean-climate zones in Western Australia, and in western Victoria and eastern South Australia (Kirkpatrick and Harris 1999b). Heathland communities also extend into areas of higher rainfall, such as on the southeast coast of Queensland ('wallum'), or lower rainfall, such as in the semi-arid zone north and west of southwest Western Australia (Fox 1995). The most significant development of heathlands can be found where there are some months in summer with a mean air temperature over 22 °C (Specht 1981d). In general, heathland is widespread but local on poor soils anywhere in Australia, especially near the coast, although not in the arid zone.

Australian heathland communities both within and without the Mediterranean-climate zone are most similar ecologically to South African fynbos communities due to low nutrient soils, floristic similarities at the family and genus level, and high levels of fire adaptation (di Castri 1991). Although Chile shares the Gondwanan heritage with Australia and South Africa, higher soil nutrients result in quite different shrubland communities. The soil nutrient difference is also evidenced in higher phosphorus levels in the leaves of Chilean matorral shrubs compared with chaparral (Margaris *et al.* 1984). Chilean and Californian Mediterranean-type communities are generally considered alike due to their younger Pleistocene assemblages, similar functioning at the ecosystem level, similar vegetation structure and their common patterns of human settlement and trade (di Castri 1991; Fox 1995).

The heathlands of the Mediterranean Basin are quite different from other heathlands of the world. These heathlands are plagioclimax communities, i.e. they are anthropogenic in origin, created by clearing forest or woodland on peaty soils, and then maintained by grazing, cutting peat, turf and gorse, and burning (Groves and Specht 1965; Gimingham 1972; Specht 1981b; Veitch *et al.* 1995). Without these human activities, the heathlands are invaded by tall shrubs and trees and succession then leads to the re-establishment of a woodland (Chapman *et al.* 1989; Webb 1990). Their low species richness and the dominant role of a few species of Ericaceae — e.g. *Calluna vulgaris* (Heather) — are additional significant differences (Gimingham 1972). However, there are some similarities with California due to evolutionary convergence and with Chile due to similar land-use patterns (di Castri 1991). Some heathlands in northern New Zealand may also have been created and maintained by human activity (Enright 1989).

The term 'heathland' as it is used in the present study refers to lowland coastal heathlands. Both the pure structural form of coastal heathland and the heathy understorey of dry sclerophyll forest are covered by this term. Heathlands that occur in

other regions of Tasmania (alpine heathlands in high altitude areas and buttongrass moorlands in the southwest of the state) differ significantly enough in their ecology to be excluded in the interest of maintaining a certain degree of uniformity. However, both wet and dry coastal lowland heathlands are included.

EDGE EFFECTS AND HABITAT FRAGMENTATION

Definitions of edges and the edge effect

A vegetation edge consists of a boundary or discontinuity between two ecological systems, vegetation types or habitats (Ranney *et al.* 1981; Gosz 1991). The adjoining habitats may be natural vegetation communities with different structural features resulting from historical or environmental influences (e.g. fire or soil boundaries) or they may be human-modified habitat types such as farmland, vegetation affected by forestry or mining, or urban areas (e.g. clearance boundaries) (Sisk and Margules 1993), or a combination of both.

In the last few decades, the concept of an edge between two vegetation types, and the effects of edges on different organisms, has changed considerably in ecological thought. Early definitions of an edge concentrated on the idea of an ecotone — the transition zone between two or more diverse communities (Odum 1959; Lidicker 1999). Ecotonal communities were described as having many of the taxa of the overlapping communities, and hence an increased variety and density of organisms — the 'edge effect' (Odum 1959). Hence, edges or ecotones were seen as a positive environment with emergent properties containing the best of both communities, for example, the increased density and diversity of wildlife populations (Sisk and Margules 1993; Lidicker 1999). A different definition of edges followed, based on the concept of island biogeography (e.g. MacArthur and Wilson (1967) and Diamond (1975)). This provided the idea that areas of terrestrial habitat surrounded by a different vegetation type or habitat could be considered an 'island' similar to a real island surrounded by water. Based on this concept of habitat islands, research and debate focused on how the size, shape, position in the landscape, distance from remaining tracts of native vegetation, time since isolation and the connectivity of the island influence species colonisation and extinction rates and species–area curves. The habitat island approach paved the way for the idea that edge effects could have a negative impact on the island of native vegetation.

Currently, research focusing on the responses of individual organisms to edges has lead to the realisation that there are few general principles regarding edge effects (Sisk and Margules 1993). Boundaries can be porous, with ecological flows moving in one or both directions, or impermeable, making the boundary more clearly defined (Wiens *et al.* 1985; Gosz 1991; Cadenasso and Pickett 2000). Boundaries can also act as a filter for disturbances

and exotic species, or provide a source for biological invasion of the interior native vegetation (Sisk and Margules 1993). Abiotic (wind, water) or biotic (animals, birds) vectors can strongly influence flows across a boundary, or the reverse can occur where boundaries impact on the movement of the vector (Wiens *et al.* 1985; Gosz 1991). It can even be difficult to define where an edge is (Murcia 1995) because a boundary can be in different places depending on the defining group of organisms (Gosz 1991; Naiman and Décamps 1991; Lidicker 1999). In general, edge effects can be positive, negative or neutral depending on the ecosystem and organism under study (Wales 1972; Sisk and Margules 1993; Murcia 1995, Debinksi and Holt 2000). The present study focuses on those edge effects that arise due to habitat fragmentation and that have a negative impact on heathlands. A negative impact is defined as a reduction in native species richness or native cover, or an increase in the species richness or cover of exotic species.

Habitat fragmentation

The massive clearing for agriculture, mining and forestry that has occurred over the last 200 years in Australia has resulted in a landscape with varying sized pockets of native vegetation, around which most or all of the original vegetation has been removed. Habitats that were, in the past, found in large continuous areas are now separated by a matrix of plantations, farmland and urban settlements (Webb and Vermaat 1990). The interactions between these fragments of native vegetation and their surroundings can take the form of abiotic or biological interactions. Abiotic or microclimate changes, such as variations in the amount or movement of radiation, wind and water, can have an important influence on what occurs at the edge (Saunders *et al.* 1991). Biological changes, such as a difference in the way the edge vegetation grows or a difference in the species present, often occur in response to a change in the edge microclimate. Microclimate and biological changes are discussed in turn below.

Changes in the microclimate

Forest edges have been the subject of recent research into microclimate differences, and the evidence from this research is useful in understanding what might happen at the edge of a heathland. Forest edges are generally warmer, brighter and drier than the interior, with higher air temperatures, higher light levels and lower humidity; they are also subject to the drying and turbulent effect of wind (Ranney *et al.* 1981; Kapos 1989; Palik and Murphy 1990; Scougall *et al.* 1993; Laurance *et al.* 2002). However, sometimes the reverse is the case, such as in a New Zealand podocarp–broadleaf forest, where photosynthetically active radiation, air temperature and vapour pressure deficit were all lower at the forest edge (Young and Mitchell 1994). The orientation of the edge can influence the presence and depth of edge effects such as increased solar radiation and wind damage. In the northern

hemisphere, edge effects tend to penetrate further into south-facing edges (Wales 1972; Palik and Murphy 1990). Southwest edges were found to have the steepest microclimatic gradients in a *Pseudotsuga menziesii* (Douglas Fir) forest in the Pacific Northwest (Chen *et al.* 1995). Similarly, south and west edges experienced the largest edge effects in forest fragments in Illinois (Gehlhausen *et al.* 2000).

Wind and water also change the conditions at the edge by carrying extra nutrients, for example pollutants carried in run-off and erosion from impervious road surfaces (Buchanan 1979; Cale and Hobbs 1991). Strong gradients in soil nutrient levels have been found at the edges of remnants, particularly from fertiliser use in adjacent farmland (Muir 1979; Hobbs and Atkins 1988; Correll 1991; Hester and Hobbs 1992; Grigg *et al.* 2000). Research into the effect of soil nutrient increases on heathlands is discussed below.

The distance that abiotic or microclimate edge effects can penetrate into forest fragments appears to vary widely, depending on the vegetation type and the edge effects under study. Laurance *et al.* (2002) give an excellent review of edge effects in a fragmented Brazilian rainforest, which range from nearly 400 m for increased wind disturbance to less than 20 m for increased vapor pressure deficit. Microclimate variables were found to have an influence up to 240 m into a *Pseudotsuga menziesii* forest (Chen *et al.* 1995). Moderate distances were measured by Young and Mitchell (1994), with gross microclimatic edge effects in a podocarp-broadleaf forest having a range of 10–100 m. Lower distances were measured by Kapos (1989) in a tropical forest in Brazil. Air temperature and humidity changes were found within 40 m from the edge and soil moisture was lower for only 20 m. Burke and Nol (1998) also found microclimate edge effects in only the first 20 m from the forest boundary. In general, temperate communities appear to be more resistant to changes from habitat fragmentation than tropical communities, due to the better dispersal ability, higher densities and wider distribution of their component species (Wilcove *et al.* 1986). Shrublands also appear to have a more closed edge, resulting in a more effective barrier to exotic seeds, fertiliser drift and soil movement (Hester and Hobbs 1992).

Changes in the vegetation

Higher light levels at the edge can lead to increased growth, higher stem densities or higher basal areas of certain species in many tropical and temperate forests (Ranney *et al.* 1981; Palik and Murphy 1990; Chen *et al.* 1992; Young and Mitchell 1994; Hansson 2000). Basal area and canopy cover can also be reduced at the edge, possibly due to windthrow (Chen *et al.* 1992). Different suites of species have been found at the edges of Amazonian rainforest (Lovejoy *et al.* 1986), podocarp-broadleaf forest in New Zealand (Young and Mitchell 1994) and temperate Australian rainforest (Fox *et al.* 1997). Often edges consist of species that are neutral to or intolerant of shade (Wales 1972; Ranney *et al.* 1981; Palik

and Murphy 1990; Chen *et al.* 1992), or that have good vegetative reproduction (Wales 1972). In contrast, some species do not respond to the edge at all (Ranney *et al.* 1981; Murcia 1995; Burke and Nol 1998). For example, Kapos (1989) found that there was no change in species composition or water stress in understorey shrubs at the edge, despite the warmer and drier conditions.

As with abiotic edge effects, the depth of the edge zone for biological effects varies widely and appears to be wider in tropical rainforests. There is some evidence that edge effects can occur over very wide spatial scales, in the order of 10 km, in some fragmented forests (Laurance 2000). However, most edge changes occur over distances of less than 1 km. In a rainforest in Queensland, Laurance (1991) found that canopy and sub-canopy damage, proliferation of lianas and invasion of non-native species could penetrate 150–500 m. A review of edge effects (Laurance *et al.* 2002) in a fragmented Brazilian rainforest show that biotic effects range from 300 m for elevated tree mortality to less than 10 m for invasion of disturbance adapted plants. In temperate forests, edge widths have been measured as 16–137 m for stocking density, growth rates and tree mortality in *Pseudotsuga menziesii* forest (Chen *et al.* 1992), 50 m for tree size and species composition in a *Banksia* woodland in Western Australia (Grigg *et al.* 2000), 10–50 m for species richness, relative cover of exotics and species composition in a mixed hardwood forest in North Carolina (Fraver 1994), 5–45 m for species composition and tree densities in sugar maple forests (Palik and Murphy 1990), 30–40 m for herbivore damage in a mesic deciduous forest in New York (Cadenasso and Pickett 2000), 20 m for species richness and abundance in a temperate rainforest (Fox *et al.* 1997), 10–20 m for density, basal area and species composition in *Quercus* spp. (Oak) forests (Wales 1972), less than 20 m for species composition in deciduous forest in Canada (Burke and Nol 1998) and 15 m for basal area and plant productivity in a Wisconsin forest (Ranney *et al.* 1981).

Information on vegetation changes at heathland edges is very sparse. Studies of the ecotones between wet and dry heathlands in New South Wales show a very diffuse boundary on the Eurunderee sand mass (Myerscough and Carolin 1986), but strong competitive interactions relating to soil characteristics and the type of roots produced between various *Banksia* species in the laboratory (Siddiqi and Carolin 1976). The edge of a *Banksia* woodland on deep sands in Western Australia experienced an influx of water, nitrate and phosphate-enriched topsoil from the adjacent farmland. Consequently, a prominent hedge of the dominant tree *Banksia prionotes* (Acorn Banksia) with an understorey of annual weeds had developed, behind which was a transition zone of smaller *B. prionotes* trees, fewer exotics and a more diverse native understorey (Grigg *et al.* 2000). Changes in species composition were found to extend 200 m from the edge of a dual carriageway through a heathland in England (Angold 1997), and 50 m from road verges through mallee in Western Australia (Cale and Hobbs 1991). Estimates of the size of an

area of heathland necessary to avoid species loss or reduce edge effects range from 588 ha (Bond *et al.* 1988) down to as small as 5 ha for maintaining habitat for certain bird species, as long as there were other remnants nearby (Martin and Catterall 2001). One researcher (Specht and Cleland 1961), suggested creating a buffer zone of approximately 400–800 m around a reserve containing heath formations to reduce edge effects in the reserve.

Heathland fragmentation

Heathlands all over the world have suffered from clearing and fragmentation along with other vegetation communities. In Australia, for example, coastal southeast Queensland lost 34% of the 1974 area of heathland by 1989 (Biodiversity Unit 1995). Chaparral is also under threat, with approximately 10% of the original native vegetation remaining (Soulé *et al.* 1992). In South Africa a similar decline is seen. In 1978 it was estimated that fynbos had declined in area by 60% of the pre-European amount, due to agriculture, forestry and urban development (Kruger 1981). Coastal fynbos is particularly at risk and is not well reserved. In some areas heathland has not suffered the same devastation, due to the undesirability of the poor soils on which it grows. For example, in the Sydney region, 85% of the pre-European woodlands and heathlands growing on sandstone remains, compared with the more fertile river-flat forests, of which only 3% remains (Benson and Howell 1990).

Much of the heathland found in Tasmania is also fragmented into small patches between more arable land which has been converted to agriculture (Kirkpatrick and Harris 1999b). The total area of coastal heathland in Tasmania has changed from approximately 425,754 ha in pre-European times to 225,754 ha in 1995 (Kirkpatrick *et al.* 1995). Clearing for agriculture (a practise dependent on phosphate fertilisers) continues to be one of the main threats to the remaining areas of coastal heathland (Kirkpatrick and Harris 1999b). This fragmentation of heathland has resulted in the remaining native vegetation consisting of a high proportion of edge habitat.

Whether a particular remnant is of any practical conservation value depends on the response of organism(s) to edges, the ratio of edge to area and how far into native vegetation edge effects penetrate (Saunders *et al.* 1987). The influence of the surrounding vegetation is also critical in determining the type and severity of edge effects — habitat 'islands' are never completely isolated (Janzen 1983). Many external factors will have an effect on the edges of the remnant, but these will have more impact where there is intensive land use (Buchanan 1979). Higher levels of human disturbance can lead to wider edge zones in remnant vegetation such as rainforest (Fox *et al.* 1997). Human-induced disturbances can therefore result in quite significant edge effects. Given the paucity of research into heathland edges and the ongoing fragmentation of heathland habitat, these

questions are particularly important for small patches of heath surrounded by highly modified habitats.

DISTURBANCE AS A FUNCTION OF EDGE IN HEATHLANDS

Edge effects in heathlands can be classified into three main types: physical, biological and human, which are often interrelated (Start 1991). Physical effects are those where changes occur due to the microclimate, and for heathlands the most critical of these are changes in nutrient cycling and the fire regime. Two physical disturbances, changes in the water relations and the impact of salt spray, are not considered in the present study. Biological effects are induced by living organisms, for example weed invasion, introduction of pathogens, grazing or herbivory. Human effects consist of direct human impacts, such as physical trampling, disturbance and cutting. Human disturbance can also have an indirect effect by influencing the physical and biological edge effects. Humans can change the nutrient cycling by deliberately or accidentally adding fertilisers, and can change the fire regime by extinguishing or lighting fires. Human disturbance can enhance the invasion of exotic species and can influence whether an invasion occurs at the edge or penetrates into the centre of a habitat fragment (Cale and Hobbs 1991). Humans may also introduce domestic stock to an area, provide improved grazing for native herbivores and increase the spread of pathogens. Each of these types of disturbance are considered edge effects in heathlands for the purposes of the present study. Each is discussed in turn, starting with the general ecology of the disturbance in the heathland ecosystem and ending with its impact as an edge effect.

Soil fertility and nutrient cycling

Heathlands mostly grow on strongly leached and acidic soils wherever they occur within and outside the Mediterranean-climate zone. These soils tend to have a very low level of total nitrogen and phosphorus, with one or more trace element also in short supply (Beadle 1954; Specht and Rayson 1957; Groves 1981a; Specht and Moll 1983). The common soil type is sandy, having developed over granite, sandstone or Quaternary sands (Specht 1981d); other soil types are brown earths, humic gleys or peat (Gimingham 1972; Groves 1981a). A heathland understorey can also develop where neutral, infertile surface soils overlay 'calcarenite soils': calcium-rich and high pH soils where elements are precipitated in a form unavailable for plant use (Specht and Moll 1983). The soils in Australian and South African Mediterranean-climate regions are the most similar, generally being deep podsolised sands (Low and Lamont 1985; Witkowski *et al.* 1990b). Kwongan soils in southwest Australia generally have even lower amounts of total nitrogen and phosphorus than fynbos (Low and Lamont 1985).

Soils under heathlands can vary at a smaller scale, and thus influence the heathland floristics. An example from Australia is kwongan, which grows on limestone, laterite and deep, leached, siliceous sands (Griffin *et al.* 1983). Limestone heaths tend to have lower species richness than laterite heaths, with more Proteaceae and Cyperaceae, and fewer Epacridaceae and Restionaceae species, than heathlands in southeastern Australia (Bridgewater and Zammit 1979; Griffin *et al.* 1983). In comparison with other heathland communities, Australian heathlands have a greater presence of nitrogen-fixing species, particularly in the early stages of succession after fire, and this influences the tight nitrogen-cycling budget (Specht *et al.* 1958; Groves 1981b). In fynbos, invasive Australian species such as those in the genus *Acacia* (Wattles) can alter the soil phosphorus status, due to large litter accumulations and rapid turnover rates (Witkowski and Mitchell 1987). Heathland soils can also vary from coastal dunes to further inland, usually with declining levels of phosphorus (Witkowski and Mitchell 1987; Adam *et al.* 1989).

Nutrient cycling

Nutrient cycling in heathlands is similar to nutrient cycling in any other vegetation type. Natural inputs of nutrients will come from the weathering of parent rock over geological time, run-on of dissolved nutrients, eroded alluvial material, the activities of animals, ash deposited after fire, microbial fixation (for nitrogen) and dust particles and aerosols carried in rainfall. Human-influenced nutrient inputs to heathlands are discussed below. Losses will occur through peat harvesting (in European heathlands), deep drainage, run-off, erosion, animal activity, loss of ash and smoke during fire and volatilisation of some elements (Groves 1981b; Groves 1983). Phosphorus is critical in delimiting plant growth; nitrogen is rarely limiting due to the presence of leguminous species (Beadle 1954). Other elements, such as sodium, can influence the heathland community — for example by separating open and closed heath on Hawkesbury Sandstone in New South Wales (Le Brocque and Buckney 1995). Different wet heath communities on Hawkesbury Sandstone in New South Wales can also be separated by phosphorus and potassium (Keith and Myerscough 1993). For excellent research into and reviews of nutrient cycling in heathlands see Chapman (1967), Groves (1981b), Specht (1981c), Margaris *et al.* (1984), Witkowski and Mitchell (1987), Chapman *et al.* (1989), Aerts (1990), and Berendse (1990).

In Australia, recent research into nutrient cycling in heathlands has shown that there is a great deal of variation in soil nutrients at the microscale. Nitrogen levels can change under the canopies of different shrub species, under a shrub canopy compared with bare soil, and under a vigorously growing shrub compared with a senescent shrub of the same species (Adams *et al.* 1994). This variation may be of the same magnitude as the variation from one season to the next or the variation before and after fire. Similar differences have been found in a fynbos community (Witkowski *et al.* 1990a). Careful measurement of indices of

available nitrogen and phosphorus in a long unburnt heathland at Wilson's Promontory, Victoria, shows that sometimes concentrations can be as high as beneath nearby eucalypt forest (Adams *et al.* 1994). In this case the accessibility of those nutrients to the heathland plants was found to be limited, due to waterlogged, anaerobic soils in winter. Nutrients can also be present but in a form that plants cannot exploit (Cheal 1996). The nutrient fluxes between different parts of the soil, between the soil and plant roots, between roots and shoots and between new and old plant tissues in heathlands are highly complex (Groves 1981b; Groves 1983).

Adaptations to low nutrients

Herbaceous plants are usually excluded by the low nutrient levels found in heathlands (see for example (Adam *et al.* 1990)). This raises the question: why are scleromorphic shrubs able to grow on such soils at all? Many heathland species have modifications to their root systems, physiology or growth rates to enable them to use nutrients unavailable to other plants. Modifications to the roots, such as such as 'proteoid' roots in the Proteaceae, ectotrophic mycorrhiza in the Myrtaceae and endotrophic mycorrhiza in Epacridaceae and Ericaceae, increase the surface area of the roots and enhance the uptake of nutrients (Groves 1981b). Australian species are particularly good at extracting nitrogen and phosphorus from the soil, as shown by the comparatively high levels of those elements in roots and aerial parts compared with fynbos plants, despite the higher fertility of the fynbos soils (Low and Lamont 1985). In comparison with plants from other vegetation types, heathland species have very low foliar nutrient levels (Specht and Moll 1983). For example, wallum species produce dry matter more efficiently, and yet have a lower phosphorus content than agricultural species, without showing any deficiency symptoms (Grundon 1972). Heathland plants can also conserve nutrients by increasing the mean residence time of nutrients in the plant. This is achieved by retaining leaves on the plant for long as possible, extracting phosphorus from senescing leaves before they fall, not dropping litter, avoiding herbivory by producing toxic chemicals in the leaves and producing a high proportion of woody stems (Specht and Groves 1966; Aerts 1990).

Nutrient-use efficiency is a common strategy for evergreen plants but results in genetically fixed or obligate slow growth rates, decreasing competitive ability in high-nutrient habitats (Grundon 1972; Aerts 1990). Although heathland vegetation grows very slowly, that growth does not appear to be nutrient-limited. The productivity of a heathland can be ten times that of an agricultural crop, such as *Trifolium subterraneum* (Sub-clover), grown on the same heathland soils (Groves 1981a). However, in absolute terms, heathland productivity is relatively low. Chaparral communities of the Mediterranean-climate regions in California have about the same productivity as heath and mallee-broombush communities in Australia, which is very low compared with a garrigue community in

southern France growing on more fertile soils (Specht 1969b). The increased growth of heathland plants when nutrient levels are raised (discussed in the section on nutrient accession below) is not necessarily proof that their growth is nutrient-limited. Adding nutrients may just allow increased growth due to better root development and better access to soil moisture (Adams *et al.* 1994).

Nutrient accession edge effects

For low nutrient systems, inputs of nutrients in any form are a type of disturbance, often occurring from the edge. Raising the fertility of heathland soils can have some dramatic effects on the heathland community. In the 1960s and 1970s in Australia, both long- and short-term field studies were conducted in which researchers deliberately added fertilisers to heathlands in various combinations and amounts. The common response by heathland shrubs or small trees was an increase in growth, often leading to the speeding up of their lifecycle (including earlier flowering) and death many years earlier than usual (Specht 1963; Groves 1965; Connor and Wilson 1967; Jones 1968; Heddle and Specht 1975; Specht *et al.* 1977). Seed germination was generally not affected, but seedling establishment could be drastically reduced (Specht 1963) and, in some cases, the density of adult shrubs was suppressed (Specht 1963; Connor and Wilson 1967). Eventually, the fertilised heathland was invaded by herbaceous native and exotic species which had a competitive advantage with their earlier growth peak and ability to access water and light (Specht 1963; Connor and Wilson 1967; Heddle and Specht 1975; Specht *et al.* 1977; Hobbs and Atkins 1991) The heathland understorey was degraded (Specht *et al.* 1977), with a decrease in species richness occurring if the overstorey cover increased (Specht and Specht 1989).

The main nutrient responsible for the changes to the heathland vegetation in the field experiments was found to be phosphorus. The dramatic decline in heath seedlings could be attributed to 'phosphorus toxicity', in fact, an imbalance between phosphorus and nitrogen (Specht 1963; Heddle and Specht 1975; Specht *et al.* 1977). Laboratory studies supported these findings, showing that high levels of phosphorus could depress yield and induce phosphorus toxicity for many heathland species, particularly in the Proteaceae family (Grundon 1972). Some species could tolerate high phosphorus if nitrogen levels were also high (e.g. *Eucalyptus pilularis* — Blackbutt) whereas other species could not tolerate high phosphorus with high nitrogen (e.g. *Banksia serrata* — Saw Banksia), or high phosphorus regardless of nitrogen levels (e.g. *Acacia suaveolens* — Sweet Wattle) (Groves and Keraitis 1976). Differences in seed size, and hence the amount of nutrients stored in the seed, are considered responsible for these different reactions (Grundon 1972; Groves and Keraitis 1976). In the field experiments, a large proportion of the phosphorus was found to be retained in the ecosystem, for 8 (Specht *et al.* 1977) or 22 years (Heddle and Specht

1975). This retention of phosphorus can then influence the N/P ratios when the next bushfire occurs (Connor and Wilson 1967; Heddle and Specht 1975).

Research into fertiliser additions in fynbos and chaparral show comparable results, with growth increases due to nitrogen and phosphorus alone or in combination depending on the species (McMaster *et al.* 1982; Witkowski *et al.* 1990a; Witkowski *et al.* 1990b). Fertilisers have often been used during the revegetation process after sand-mining in Australia. The results have been increased growth or biomass of native vegetation leading to earlier and more severe competition, also decreasing species density and polydominance (Clark 1975; Thatcher and Westman 1975). Even European heathlands, despite their ecological differences, can be invaded by herbaceous species when nutrient levels (particularly nitrogen) increase (Heil and Diemont 1983; Aerts and Berendse 1988; Aerts *et al.* 1990).

What all this literature has in common is a focus on the effects of the addition of large amounts of nutrients to heathlands. The amounts that have been used generally aim to raise the nutrient levels to those found in agricultural systems or in more fertile native communities. There is a lack of evidence for any effects resulting from the addition of smaller amounts of fertilisers or other nutrients. The exception is research into the 'ashbed' effect, that is, additions of nutrients to heathlands after fire. This can result in increases in shoot size, number of inflorescences and above-ground biomass in chaparral (McMaster *et al.* 1982) and increases in shoot length, dry mass and mean leaf area in fynbos (Witkowski *et al.* 1990a; Witkowski *et al.* 1990b). However, there are many other situations, particularly in cleared and fragmented heathlands, where small amounts of nutrients from human activities might be added to the heathland.

Human-induced nutrient accessions can occur due to drift from aerial top-dressing of improved pasture or forest plantations (Muir 1979; Specht 1981a), the use of fire-fighting foams or retardants (Larson and Duncan 1982; Bradstock *et al.* 1987; Marsden-Smedley 1992), the application of fertilisers during rehabilitation after sand-mining (Clark 1975; Thatcher and Westman 1975), erosion or run-off from agricultural areas (Grigg *et al.* 2000), irrigation using sewage effluent (Dutch and Wolstenholme 1994), run-off from urban areas including septic tank overflow and leaching from disposal sites (Clements 1983; Lambert and Turner 1987), litter and excreta from recreational use (Specht 1981a), dust from road-building material (Tamm and Troedsson 1955; Clemens and Franklin 1980), and nitrogen from vehicle exhaust (Angold 1997). Many of these nutrient accessions have been poorly studied (such as the effects of sewage effluent) or have not received attention in the Australian context (such as the effects of fire-fighting foams). Increased nutrients, particularly in conjunction with soil disturbance, can enhance the ability of exotic species to invade at the expense of native species (Hobbs and Atkins 1988).

Fire ecology and impacts

Pyric succession and species richness

Fire is very important to heathland ecology, and needs to occur periodically to maintain the structure and species richness of the heathland vegetation. The flammability of heath species is increased by the large amounts of resins, tannins and essential oils in their leaves (Specht 1981b). Pyric succession has been much studied in heathlands and varies between the different Mediterranean-climate zones. Australian heathlands, fynbos and chaparral all experience plant recruitment immediately after burning, have fire-stimulated germination in shrubs and tend to stagnate in the absence of fire (Gill and Groves 1981). Australian heathland and fynbos have more geophytes and persistent herbaceous perennials but fewer dominant annuals and short-lived fire perennials than chaparral (Gill and Groves 1981).

Generally after a fire the growth curve of Australian heathland species shows a rapid and exponential increase in biomass, with productivity then slowly decreasing over time (Specht *et al.* 1958; Jones *et al.* 1969). In some cases there may be two biomass peaks, such as at Dark Island Soak in South Australia. The first peak occurs in the period directly after the fire when short-lived perennial species germinate from seed and other species resprout from rootstock. The second peak occurs when the short-lived perennials die and the long-lived perennials that have germinated from seed (e.g. *Banksia ornata* — Desert Banksia) have their growth spurt, after approximately 10 years (Specht *et al.* 1958; Groves and Specht 1965). In chaparral, temporary cover (annual and perennial herbs, short-lived undershrubs) reaches a peak of growth 2–5 years after the fire and disappears as the permanent cover replaces it (larger scleromorphic trees or shrubs) (Specht 1969a). Chaparral reaches its peak above-ground biomass about 25 years after fire (McMaster *et al.* 1982).

Species richness is usually directly related to age since fire, declining over time in Australian heathlands (Specht *et al.* 1958; Russell and Parsons 1978; Clemens and Franklin 1980; Specht and Specht 1989), in chaparral and in fynbos (Gill and Groves 1981). In the Dark Island heath in South Australia, richness declined by 44% in 25 years (Specht *et al.* 1958), whereas at Wilson's Promontory in Victoria it declined by 22% in 21 years (Russell and Parsons 1978). Also at Wilson's Promontory, although a species decline was measured, no major floristic change with time since fire was detected (Russell and Parsons 1978). In chaparral, the highest species richness occurs in the first year post-fire, because generalist perennials, generalist annuals, fire-tolerant annuals and fire-tolerant perennials are all present at the same time (Keeley *et al.* 1981). Species loss in chaparral is more rapid when compared with Australian heathlands and fynbos (Kruger 1983). There are two main

reasons for the decline in richness post-fire: firstly, the rapid disappearance of temporary fire ephemerals; secondly, the gradual suppression of understorey species by overstorey species (Specht *et al.* 1958; Kruger 1983). For example, species richness was found to overtake pre-fire levels after one year and peak at four years in a coastal heathland in Victoria (Posamentier *et al.* 1981). Many of these species were not found before the fire, but would probably be out-competed as the large shrubs regain dominance. In contrast to the general trend of decreasing species richness, Purdie (1977a) found that, in a heathy understorey in the ACT, the floristic composition was the same after a fire. However, the relative dominance of species was quite different, reflecting the different regeneration strategies of the different plants. High fire frequency was also found not to affect species richness in a heathland near Sydney, New South Wales (Bradstock *et al.* 1997).

Resprouting and reseeding

There are two main strategies with regard to post-fire regeneration: to resprout from underground organs or to regenerate from seed. Many plants prefer one strategy over the other, although some plants (e.g. *Banksia aspleniifolia* — Fern Leaf Banksia and *Banksia marginata* — Honeysuckle or Silver Banksia) can do both (Siddiqi *et al.* 1976a). In chaparral, fynbos and Australian heathlands both strategies are common (Specht 1969a; Gill and Groves 1981), although resprouters appear to dominate in many Australian heathlands (Russell and Parsons 1978; Kruger 1983). The balance of the two strategies in any one heathland community will depend on the frequency and intensity of the fire regime, particularly on the characteristics of the most recent fire.

Obligate resprouters are plants that mainly regenerate after fire by resprouting from underground organs such as lignotubers, horizontal root systems, prostrate branches or rhizomes. Most resprouters also produce seed. Resprouters are found in heathlands in Europe, and include *Calluna vulgaris* (Heather) in British heathlands and *Quercus coccifera* (Kermes Oak) in garrigue vegetation (Specht 1969a; Mallik and Gimingham 1983). Dominant resprouters in chaparral, such as *Adenostoma fasciculatum* (Chamise), are not confined to resprouting, and vary in their survival of fires and their resprouting and seeding habits (Kruger 1983). In contrast, resprouters in Australian and South African heathlands tend to have high survival rates and are more likely to be obligate resprouters (Kruger 1983). The relative dominance of resprouters in the community is not just influenced by the fire frequency. The position of the dormant buds (Purdie 1977a), whether it is a wet or a dry heath (Myerscough *et al.* 1995) and the presence of an overstorey (Keith and Bradstock 1994) all affect the prevalence of resprouters.

Obligate seeders do not regenerate vegetatively after fire and in fact may only germinate after other vegetation has established. Their seeds are often stored in woody fruits in the

canopy or will only open after fire, needing heat, chemicals or ash to break dormancy. Serotiny (canopy-stored seed) not only cues the seed release precisely to the availability of a post-fire seedbed, but also protects the seeds from predation in the periods between fires (Gill 1981). Fire also removes other inhibitory factors such as seedling predators, shading, and toxins or pathogens in the soil or litter (Purdie 1977b). The relative dominance of obligate seeding species is influenced by the effect of temperature on germination (Enright and Lamont 1989), the time since fire (Purdie 1977b), the dispersal ability of the seed (Hammill *et al.* 1998) and the seed storage mechanism. Canopy-stored seed is easily exhausted after fire whereas soil-stored seeds are more likely to survive a series of frequent fires (Bond 1984; Bradstock *et al.* 1997).

Fire regime

Fire frequency is probably the best researched element of the fire regime, other elements being fire intensity and fire season. Fire frequency is highly variable in all Mediterranean-type ecosystems. In Australia, fire frequencies have been measured at approximately 25 years for Dark Island Soak heathland (Specht *et al.* 1958), 20 years for Wilson's Promontory (Russell and Parsons 1978), 12–15 years in New South Wales (Siddiqi *et al.* 1976a), and 5–7 years for dry sclerophyll forest with a heathy understorey in Western Australia (Christensen and Kimber 1975). They can also occur at longer or shorter intervals than these. In fynbos the frequency is more commonly 20–40 years and for chaparral 35–90 years (Kruger 1983). Although time since fire explains a large amount of the floristic variation, the average inter-fire interval and the variability of the inter-fire interval are also important (Morrison *et al.* 1995)

Different heath species have different fire frequency requirements for persistence. *Angophora hispida* (Dwarf or Scrub Apple) requires at least 9 years between fires to enable juveniles to be recruited into the population (Auld 1990). Woody species in scrub-heath in New South Wales need intervals of more than 7–8 years (Morrison *et al.* 1996), *Banksia serrata* (Saw Banksia) needs 6 years, *Isopogon anemonifolius* (Broadleaf Drumstick) needs 13 years (Bradstock and Myerscough 1988), endemic kwongan *Banksia* species prefer 10 year-intervals (Cowling *et al.* 1990) and *Banksia grandis* (Bull Banksia) up to 35 years between fires (Abbott 1985). At the other end of the scale, maximum seeding in *Xanthorrhoea australis* (Austral Grasstree) would be promoted by six-year fire intervals (Gill and Ingwersen 1976). Australian heathland and South African fynbos do generally not accumulate enough standing vegetation and litter to sustain a fire more frequently than every 4–5 years (Specht *et al.* 1958; Kruger 1983). Regular fire intervals may actually not be the most appropriate approach to conserving species richness — a varied fire interval can maximise the conservation of heath species (Keith and Bradstock 1994).

The effects of changes in the fire regime

Fire has become an important management tool for many vegetation types including heathland. Burns can be prescribed for many reasons, including habitat management, fuel reduction and species conservation. A conflict can arise between the fire frequency needed for fuel hazard reduction (shorter) and the fire frequency needed for species conservation (longer) (Morrison *et al.* 1996). Fires can impact heathlands from road, track or fire-break edges. When fires are deliberately lit, people have control of the fire frequency, the fire season and the fire intensity. Changes in the fire regime can have a big influence on the heathland community.

When fires occur too often, obligate seeders will not have enough time to mature and set seed, thus reducing or eliminating these species from the stand (Specht 1981a). This has been demonstrated in three different heathlands in New South Wales by Siddiqi *et al.* (1976a), Nieuwenhuis (1987) and Bradstock *et al.* (1997). Some obligate seeders are affected more negatively than others — worst affected are serotinous, wind-dispersed, heavy seeded shrubs of about 3–4 m in height (Nieuwenhuis 1987). Approximately 4–8 years after a fire is the time of peak reproductive activity for heath species in Australia, therefore obligate seeders may need a minimum inter-fire interval of 8 years (McFarland 1990). Too frequent fires can also lead to dominance by non-shrub species, such as *Lomandra longifolia* (Sagg) (Kirkpatrick 1977) or *Pteridium esculentum* (Austral Bracken) (Duncan and Duncan 1995) in Tasmanian heathland, grasses and annuals in Western Australian woodlands (Christensen and Kimber 1975), and perennial grasses or understorey graminoids in fynbos (Kruger 1983). Herbaceous resprouters are well able to cope with frequent fire intervals (less than five years) (Bradstock *et al.* 1997). Sedges are particularly likely to dominate when fires become too frequent, using their large carbohydrate reserves to recover rapidly (Kirkpatrick and Harris 1999b). It has been suggested that *Xanthorrhoea* (Grasstree) species would become more dominant in frequently burnt communities, because of the increase in flowering that fire causes in these species (Gill and Ingwersen 1976). However, in more recent research, Curtis (1998) found that more frequent fires were detrimental to *Xanthorrhoea australis* (Austral Grasstree), causing more frequent deaths from stem fracture compared with unburnt sites.

A lack of fire can lead to the gradual decline in species richness and productivity of the heathland. Over time, a few shrub species may come to dominate, resulting in a tall, thick species-poor community developing which, once over 2 m, would become 'scrub' and no longer be classified as a heathland. (Specht *et al.* 1958; Clemens and Franklin 1980; Specht and Specht 1989). Tall shrubs from outside the heathland can invade in the absence of fire, for example *Leptospermum laevigatum* (Coastal Teatree) in Victorian heathlands (Molnar *et al.* 1989) and a number of *Leptospermum* species in Tasmania (Kirkpatrick and

Harris 1999b). Some heathland species may not be able to persist as seed or underground storage organs during long fire-free intervals, becoming locally extinct (Specht *et al.* 1958). Projective cover of resprouters decreased after long fire intervals (once or twice in 20 years) in a heathland in New South Wales, perhaps due to the reduction in sprouting ability or competition from obligate seeders (Nieuwenhuis 1987). Lower fire frequency in fragmented islands of fynbos has resulted in more graminoids, fewer geophytes which flower after fire and fewer resprouters (Bond *et al.* 1988). Chaparral communities are unable to regenerate in the absence of fire — under the even-aged mature canopy there are few shrub species and little herbaceous growth (Keeley *et al.* 1981). In contrast, prolonged lack of fire can lead to a type of cyclic succession where gaps occurring due to shrub senescence are recolonised by dominant shrubs (Specht *et al.* 1958), or allow less dominant species to become dominant, e.g. in chaparral (Patric and Hanes 1964).

The season of the burn can affect regeneration, for example fynbos experiences successful regeneration of seedlings after summer or autumn fires but loss of seedling populations after winter or spring fires (Kruger 1983). In an Australian heathland, recruitment of some species is more reliable after winter or spring burns , due to more predictable rainfall compared with after summer burns (Bradstock and Myerscough 1981). Differences in recruitment in different fire seasons can be explained by seed predation. Bond (1984) found that the germination cue for serotinous Proteaceae in fynbos only occurred in autumn, so winter or spring fires resulted in a delay between seed release after fire and the germination cue.

Fire temperature is linked to the fire season and also has an effect. When fires are hotter, lignotubers are more likely to be killed, litter (containing toxic substances that stop germination) is more likely to be removed and hard-coated seeds are more likely to have their seed coats cracked. The result is that germination from seed is easier but resprouting is harder (Posamentier *et al.* 1981). Fire temperature also affects the flowering of some plants, such as *Xanthorrhoea* (Grasstree) species (Lamont *et al.* 2000). Higher temperatures and drier soils cause higher amounts of nitrogen volatilisation (DeBano *et al.* 1979; Dunn *et al.* 1979; Rundel 1983), so fire temperature will influence nutrient cycling.

Nutrients and fire

The regular fires that are part of heathland ecology significantly affect nutrient cycling. Immediately after a fire, some phosphorus is lost in the smoke but much is returned to the soil surface in the ash. The temporary increase in phosphorus usually returns to pre-fire levels within one year (Siddiqi *et al.* 1976a; Brown and Mitchell 1986; Adams *et al.* 1994), due to rapid leaching from the soil. Nitrogen is volatilised and lost (Loneragan and Loneragan 1964; Groves 1981b); in European heathlands the losses can be as high as 95%

(Chapman 1967). Nitrogen losses can be partially offset by the growth of the many nitrogen-fixing species in Australian heathlands, which can produce enough nitrogen in the short-term to balance the excess phosphorus (Heddle and Specht 1975). In European heathlands, nitrogen is soon returned from the ions in rain (Groves 1981b), and in chaparral from both nitrogen-fixing plants and free-living nitrogen-fixing microbes (Rundel 1983). In the first 15 years after fire, heathland plants have a rapid uptake of nutrients, which then slows and is replaced by increased recycling in the 15–50 year period (Specht *et al.* 1958). Nutrients steadily accumulate in underground organs over time since fire and are therefore stored for future survival rather than used for current growth (Specht *et al.* 1958). Repeated burning reduces the levels of carbon, nitrogen, potentially mineralisable nitrogen and phosphatase activity but increases the levels of phosphorus in the top 2 cm of soil (Adams *et al.* 1994). Frequent fires or intense rainfall soon after a fire (or both) can deplete the nutrient status of the heathland (Groves 1983). Fire has generally only been accidentally included in research on the effects of nutrient additions, and few experiments have deliberately investigated the interaction between fire and fertilisers.

Community composition, weed invasion and pathogens

Species richness and community structure

In Australia, the common families of Epacridaceae, Proteaceae and Myrtaceae dominate the heathland community. Proteaceae, in particular, can contribute 32–84% of the aerial biomass (Specht *et al.* 1958; Low and Lamont 1990). In South Africa, Proteaceae can be even more abundant, reaching a maximum of 98% in a fynbos community in the Western Cape (Low and Lamont 1986). Although heathlands consist primarily of evergreen scleromorphic shrubs, monocotyledonous species are often very important. These monocotyledons will usually have evergreen scleromorphic leaves themselves — herbaceous seasonal grasses are uncommon (Specht 1981b). Families represented in Australia are Cyperaceae, Liliaceae, Restionaceae and Xanthorrhoeaceae (Specht 1981b). Restionaceae species are also a characteristic feature of South African fynbos (Taylor 1978 in Low and Lamont 1990). These species become increasingly important in wet heathlands, producing a community that could be defined as a 'graminoid heathland' (Specht 1981b).

Two common features of heathland floristics are the high species richness and the lack of dominance of any one species. Species from different genera and families can have similar growth forms and structures (Specht 1981a; Adam *et al.* 1990). Where there are many species present within one genus, microclimate segregation may occur due to differing physiological tolerances, such as needing soil disturbance to germinate or a higher need for water (Clarke *et al.* 1996). Species richness per unit area in the heathy understorey

declines as the overstorey cover or density increases (Specht and Specht 1989; Keith and Bradstock 1994).

Within the Mediterranean-type climate regions of the world, parts of Western Australia and the Cape region of South Africa have counts of species richness approaching that found in tropical rainforest. These two areas are followed by Chile, California and the Mediterranean Basin in order of richness (di Castri 1991). Even though richness in the Mediterranean Basin is lower than that of similar climate zones elsewhere, it is still much higher than other European regions. Of the two Mediterranean-type regions in Australia, shrubland and heathland communities in Western Australia are floristically more diverse than those of southeastern Australia. Western kwongan species richness is also much higher than in eastern heathlands outside the Mediterranean-type climate region (Fox 1995). Heathlands in subtropical regions generally have lower species richness than southern heathlands (Dwyer *et al.* 1981).

The high species richness in fynbos and kwongan is generally attributed to the extremely poor soils (Bond 1983), the location within the Mediterranean-climate zone and time to evolve (Specht 1981d). Various Australian studies support the 'resource competition model' of species richness. Major trends in floristic composition in wet heath in New South Wales were related to soil nutrients (Keith and Myerscough 1993), a significant non-linear inverse relationship between soil total phosphorus and species richness was measured by Adam *et al.* (1989), and Hahs *et al.* (1999) found a relationship between higher species richness and lower nutrient levels. High beta diversity, that is a very patchy environment, plus high habitat specificity are other reasons suggested for the species richness of heathlands and shrublands in Mediterranean ecosystems (Cody 1986). Burgman (1988) found spatial patchiness (rather than soil characteristics) to be a significant factor in kwongan community species richness.

Weed invasion

One of the most insidious edge effects for heathlands is the invasion of exotic species from adjacent farmland, roads or suburban areas. Disturbances (such as fire, increasing nutrients and soil disturbance) are generally considered to increase the invasibility of natural communities (Fox and Fox 1986; Hobbs 1989; Rejmánek 1989), particularly if they increase the availability of a limiting resource (Hobbs 1989). The disturbance of fire, a fundamental part of heathland ecology, does not necessarily increase heathland invasibility. Hobbs and Atkins (1991) and Hester and Hobbs (1992) found that fire did not have any effect on colonisation by exotic species in shrublands and heathlands in Western Australia.

However, exotic species are able to invade when a change in fire frequency, or a fire itself, is combined with soil disturbance or the addition of nutrients. At a small urban reserve in Victoria, the association of a recent fire with rubbish dumping, trampling and soil disturbance resulted in a high density of exotic seedlings which then suppressed the regeneration of native species (Molnar *et al.* 1989). In another example, increasing fire frequency and the input of nutrient-laden dust was found to allow exotic grasses to invade a closed heathland in Western Australia (Bridgewater and Backshall 1981). Roads provide another source of disturbance which, when combined with burning, can result in increased weeds (Milberg and Lamont 1995), particularly annuals (Hobbs and Atkins 1991). Fire and nutrients can also combine to allow a heathland to be invaded by troublesome native species that change the vegetation structure and decrease the species richness, such as *Leptospermum laevigatum* (Coastal Teatree) in Victoria. Where there is an increase in soil fertility, the mycrorrhizal inoculation of *L. laevigatum* roots increases and the shrub can invade and, if the fire frequency is low, persist (Specht 1981a). Both this shrub and *Kunzea ambigua* (Tick Bush) can then build up foliage and litter nutrients to a far higher level than that which occurs in adjacent uninvaded heath (D. Cheal, School of Botany, University of Melbourne, *pers. comm.* 1996). Lack of fire and physical disturbance from people are similarly implicated in the invasion of *Pittosporum undulatum* (Sweet Pittosporum or Victorian Box) into heathy woodlands in New South Wales (Rose 1997).

Nutrient additions alone can be enough of a disturbance to allow weed invasion into heathlands, as seen in the fertiliser experiments described above. Suburban areas surrounding bushland provide both increased nutrients and many sources of exotic propagules. In one such situation near Sydney, New South Wales, heathy vegetation was found to be decreasing and wet sclerophyll, rainforest and weedy communities were all increasing (Clements 1983). In other studies near Sydney, those species that were increasing, particularly exotics, had higher concentrations of nutrients in their foliage (Lambert and Turner 1987). Changes in street drainage, human waste disposal and dumping of grass clippings (resulting in more nutrients) also lead to the rapid increase of three common exotic species (Buchanan 1979). In the Netherlands, perennial grasses can invade heathland after an increase in nutrient supply, if light is not limiting (Berendse 1990). Changes in nutrient cycling due to particular plants, for example *Santalum spicatum* (Sandalwood) can provide enough of a disturbance to increase the biomass of exotic annuals (Hobbs and Atkins 1991). An increase in phosphorus in itself is adequate to increase exotic species richness (Adam *et al.* 1989), as is raising nitrogen levels, such as from fertiliser drift, stubble drift and increased plant litter (Hester and Hobbs 1992).

Nutrient increases combined with other types of disturbance provide even more ideal conditions for exploitation by exotics. Weed establishment is increased when soil disturbance is present, but weed persistence is increased to a greater extent when nutrient

additions are added to soil disturbance (Hobbs and Atkins 1988). Soil disturbance was most important for the establishment of the deliberately sown exotic grass *Avena fatua* (Wild Oats) in heathland, but fertiliser addition was most important for biomass production after the initial establishment (Hobbs 1989). As discussed for fire and disturbance, roadways also provide a combination of physical disturbance and increased nutrients — from both the building and subsequent use of the road. Roads supply a constant seed rain, regular disturbance such as drainage control, cutting of vegetation and road widening, and increasing water run-off and nutrient levels (Milberg and Lamont 1995). The grass *Molinia caerulea* (Purple Moor Grass) increased and *Calluna vulgaris* (Heather) decreased in abundance next to a dual carriageway through a heathland in England, thought to be due to eutrophication from vehicle exhausts, particularly nitrogen (Angold 1997). Roads through a mallee community in Western Australia resulted in weed invasion, which was positively correlated with increased phosphorus and nitrogen levels (Cale and Hobbs 1991). Increased weed invasion into wet and dry sclerophyll forest also occurred along a roadside in Victoria (Amor and Stevens 1975).

Disturbance without nutrient additions can also be sufficient to enhance weed invasion. Exotic grasses were able to establish in a coastal heathland in New South Wales subject to physical disturbance from visitor pressure (Clemens and Franklin 1980). There is also some evidence to suggest that one particular type of disturbance — destruction of the canopy — can make a big difference to the invasibility of heathland communities. Exotic species are generally more abundant at the edges of native vegetation remnants, but their penetration into the centre can be hindered by the closed edge of a heathland compared with a more open woodland canopy (Hobbs and Atkins 1988; Hester and Hobbs 1992). Sometimes invasion can occur without any disturbance at all, as in the case of a fire-adapted Australian native (*Acacia saligna* — Golden Wreath or Orange Wattle) invading South African fynbos communities (Holmes 2002).

Phytophthora cinnamomi

The root pathogen *Phytophthora cinnamomi* (Cinnamon fungus) is one of the greatest threats to the conservation of Australian coastal heathland ecosystems (Malajczuk and Glenn 1981; Kirkpatrick and Harris 1999b). This soil-borne fungus causes death by nutrient starvation and/or droughting, by attacking the root system of susceptible plants (Malajczuk and Glenn 1981). The fungus is found in all continents of the world, but there is some debate about whether it is a recent introduction to Australia (Weste 1974; Podger 1989; Podger and Brown 1989) or is a long-term resident (Pratt and Heather 1973). There is general agreement that the fungus has been relatively newly introduced to Tasmania (Podger and Brown 1989) and Western Australia (Weste 1974; Malajczuk and Glenn 1981). Other species of *Phytophthora*, e.g. *P. citricola* in Western Australia (Shearer and

Tippett 1989) and various *Phytophthora* species in fynbos (Davis 1992), are also potential problems.

Conditions that favour the fungus are shallow soils with poor drainage, and wet periods with temperatures above 12 °C followed by subsequent periods of water stress (Weste 1974). In Tasmania, the critical conditions are mean temperatures greater than 7.5 °C and a mean annual rainfall of greater than 600 m (Podger *et al.* 1990a). Fire has the potential to reduce populations of the fungus, as it is not tolerant of very high temperatures (Podger 1990). However, this potential is usually not realised in the field due to refugia for the fungus deep within the soil or sheltered beneath rocks, and the massive regeneration after fire of the very species that are most susceptible to the disease. The fungus spreads actively downhill with drainage water and more slowly uphill in the soil or by growing through root systems of susceptible plants (Specht 1981a; Shearer and Tippett 1989). Most initial infestations originate from roads and tracks (Weste and Law 1973; Podger and Brown 1989), and the passive spread by transportation of infected soil, vehicles, tools, boots or animals appears likely (Weste and Law 1973; Podger *et al.* 1990a)

P. cinnamomi predominantly attacks woody plants (Podger and Brown 1989; Wills 1993), and affects a number of families that are either dominant or co-dominant in the heathland community (Weste and Law 1973). The most susceptible families in Australia are Epacridaceae, Fabaceae, Myrtaceae, Proteaceae and Xanthorrhoeaceae (Malajczuk and Glenn 1981). Many of the genera in Ericaceae in the Northern Hemisphere are also susceptible to the fungus (Malajczuk and Glenn 1981). There is a big variation in susceptibility within members not only of the same family but also the same genus (Malajczuk and Glenn 1981; Podger 1990; Wills 1993; Barker and Wardlaw 1995). Monocotyledons are generally resistant (Podger 1989), particularly Restionaceae, Cyperaceae (Malajczuk and Glenn 1981; Wills 1993) and Poaceae (Shearer and Tippett 1989), and also most ferns (Podger and Brown 1989).

The effects on heathlands can be dramatic and devastating. Because the fungus attacks roots, deaths often occur during periods of water stress, leading to an increase in the available soil water which consequently favours the pathogen (Weste 1974). Once the disease is established there can be a rapid decline in shrub species, floristic impoverishment of the heathland and an increase in sedges and rushes (Weste and Law 1973; Weste 1974; Wills 1993). The structural features of the community can change completely, such as from a shrubby forest to a sedgy open woodland (Duncan and Keane 1996). Host plants intolerant to waterlogging and susceptible to *P. cinnamomi* will become quickly absent from the community (Weste and Law 1973). In some cases almost complete local extinction will occur, such as *Xanthorrhoea australis* (Austral Grasstree) from heathlands on Cape Barren Island in Tasmania (Kirkpatrick and Harris 1999b) and from dry sclerophyll forest in Victoria (Duncan and Keane 1996). Disease activity can decline in

the soil under individual dead and dying plants, over a time period as short as 2 years (Duncan and Keane 1996). Decline in the potential pathogen activity over the whole site occurs over a longer time period. Weste and Kennedy (1997) found that, 20 years after the initial infestation, two thirds of the susceptible species were able to recolonise the previously diseased area.

Grazing

Grazing is not generally considered a major threat to Australian heathlands due to the lack of palatable heath species and the density of the vegetation (Specht 1981a; Hobbs 1991), but can still be a problem in relation to fire, weed invasion and in terms of trampling. Native animals will generally use the heathland for cover and graze in more fertile grasslands if available. After fire, both native and feral animals will seek out the regenerating herbaceous species and the new shoots of heath species, before they become woody and unpalatable (Specht 1981a). In dry sclerophyll forests a reciprocal relationship can occur, where seedlings and resprouts of plants that become unpalatable without grazing are kept palatable by the presence of grazing (Dickinson and Kirkpatrick 1986). Heavy grazing after burning does have the potential to dramatically change the community floristics and structural composition, reducing the biomass, cover and reproductive capacity of any palatable species (Leigh and Holgate 1979). One potential negative effect of stock grazing is the selective grazing of palatable native species that may be unable to recover from defoliation, and the avoidance of some exotic species due to unpalatable prickles and thorns, thus encouraging weed invasion (Amor and Piggin 1977). Other potentially destructive effects of domestic stock grazing arise from trampling and the addition of nutrients from faeces, which can result in significant changes to vegetation structure and composition and to a range of soil properties (Hobbs 2001). Weed invasion into woodlands in Western Australia (Hobbs 1991; Pettit *et al.* 1995; Yates *et al.* 2000) and into temperate rainforest in New South Wales (Fox *et al.* 1997) has increased from such effects of stock grazing. In the plagioclimax European heathlands, the presence or absence of grazing has been shown to result in the formation or loss of heathlands. The presence of sheep grazing in the Massif Central in France allowed heathlands to develop due to significant impoverishment of the soils (Coquillard 1995), and the lack of grazing by rabbits has lead to many *Calluna vulgaris* (Heather) heathlands in England becoming scrub or woodland (Chapman *et al.* 1989).

Physical disturbance and clearing

Heathland vegetation is easily damaged by physical disturbance such as tracks and trails, vehicle movements and trampling (Rose and Webb 1994). Examples of the effects on heathland come from widely varying studies in different communities in the

Mediterranean-type climate zone. The edges of Californian chaparral are pushed back due to trails, expanding gardens and destruction of plants by people. The brittleness and slow growth rates of the vegetation result in very slow recovery from such physical disturbance (Soulé *et al.* 1992). Tank tracks in shrub steppe in California have the effect of increasing the amount of bare ground, litter and exotic annuals and decreasing the cover of perennial native grasses and shrubs. These effects increase the potential for erosion and fire frequency, and decrease infiltration and nutrient cycling (Watts 1998). Cutting and covering plants with roadway ballast in heathland in England resulted in the presence of invasive species but no loss of native species (Rose and Webb 1994). Adverse impacts (such as crushing of vegetation, soil erosion and compaction, damage to tree trunks and smothering of plants with soil) occurred from trail bike riding and developmental works in heathy woodlands in New South Wales (Buchanan 1979). In South Australia, heathland that had been 'rolled' with a bulldozer to create fire trails suffered no loss of species or weed invasions but did experience changed soil conditions, a reduction in biomass and alterations to community composition (Pelton and Conran 2002).

Clearing obviously also damages the heathland community, as already discussed in the section on habitat fragmentation. Sand mining is very disruptive to Australian heathlands, firstly, because of vegetation removal, soil disturbance and stockpiling and, secondly, because revegetation attempts (until recently) usually included fertilisers. The entire germination environment changes (light, moisture, temperature, nutrients), and the seed store will be disrupted, seed stored in the soil might not be viable after soil stockpiling and canopy-stored seed may be lost entirely (Clark 1975). Clearing and earthworks in a coastal heathland in New South Wales lead to erosion of plants and topsoil from sealed and compacted surfaces (Clemens and Franklin 1980). Tillage in fynbos communities cleared by fire resulted in a change in the reflectivity and moisture of the soil and suppressed post-fire establishment of fynbos species (Davis 1992). Clearing heathland in Southern Britain resulted in soils with lower organic content, mixing of the previously distinct soil layers, changes in nitrogen mineralisation and higher levels of phosphorus due to regular fertilising (Pywell *et al.* 1994). The removal of smaller amounts of vegetation, such as picking the heads of proteaceous species for the wildflower industry or for seed collection can also cause problems. Such flower heads contain relatively high levels of nutrients and their removal, particularly in combination with fire, could result in significant losses of nutrients and the reduction in seedling establishment in fynbos and kwongan ecosystems (Low and Lamont 1986; Maze and Bond 1996).

RESEARCH QUESTIONS AND THESIS STRUCTURE

Research questions and null hypotheses

Much of the literature on conservation of fragmented Australian heathlands reviewed in this chapter suggests that they are particularly susceptible to edge effects, especially those related to nutrient accession. The addition of large amounts of fertilisers in field and laboratory experiments has resulted in large changes to heathland structure and floristics over time. Most notably, the condition of the heathland is reduced due to the invasion of exotic species that have a competitive advantage in the high nutrient environment. Weed invasion into other plant communities is generally facilitated by any type of disturbance. This evidence, combined with the evidence for negative impacts to heathlands after nutrient additions, has lead to an assumption that any addition of nutrients to heathland will result in serious degradation of the condition of the heathland community. This approach results in the view that small patches of remnant heathland surrounded by agricultural land would not have any conservation value as they would be overwhelmed by edge effects.

This view has not been tested by experiment or observation. Thus, this thesis addresses the following questions:

- In what circumstances are heathlands susceptible to edge effects?
- What interactions are there between low levels of nutrient additions and other types of disturbance? In particular, do fire and nutrient accessions have interactive deleterious effects?
- Are small areas of heath worth conserving or will they be overwhelmed by edge effects?
- How does human management influence heathland edge effects?

In this thesis, the different field experiments are designed to mimic realistic nutrient additions to heathlands, in the form of sewage wastewater, fertilisers and fire-fighting foam. These nutrient disturbances are combined with other types of disturbance such as fire and weed invasion in order to gain an understanding of how such disturbances affect heathland condition. Widespread observation of the boundaries between heathland and cleared land was undertaken to determine what management practices help small heathland remnants to survive. Thus the null hypotheses addressed in the present study are: 1) realistic applications of nutrients to heathlands have no effect on their species richness, composition and growth; 2) realistic applications of nutrients to heathlands, in interaction with fire, have no effect on their species richness, composition and growth; and

3) the nature of the disturbance regime on both sides of the heath–pasture boundary has no effects on heathland species richness, composition and growth.

Thesis structure

This introductory chapter has presented and described heathland, introduced the concept of edge effects and defined which edge effects occur in heathlands in Australia and other parts of the world. Chapter 2 discusses a study monitoring the effects of irrigation with sewage water on a coastal heathy woodland at Stieglitz on the east coast of Tasmania. This chapter combines two different lines of ecological research, namely, soil, nutrient and plant relations in heathland communities, and the effect of using land-based systems for the disposal and re-use of wastewater. The aim was to monitor any changes in the plant species composition, the amount and proportion of different ground covers and to investigate whether common species react to sewage irrigation.

Chapter 3 describes a manipulative field experiment where the interactive effects of fire, fertilisers and weed seeds on coastal heathland are investigated. The study site is an area of pure heathland within the Peter Murrell Nature Reserve south of Hobart. There were three main aims of the experiment, firstly, to find out if small amounts of nutrients have any effect on heathland vegetation. Secondly, to include fire by controlling the timing of both the burn and the nutrient additions, and investigate the interactive effects of fire and fertilisers. Thirdly, to investigate the interactions between the addition of fertilisers, the disturbance caused by fire, and weed invasion in heathland.

Chapter 4 deals with another manipulative field experiment investigating the interactive effects of fire and fire-fighting foam. The study site is an area of pure heathland within the Mount William National Park in the far northeast of the state. Fire-fighting foams are regularly used in fire management but their ecological impacts have not been adequately assessed. Therefore, the aim of this experiment was to measure the effects of fire-fighting foam, with and without burning, on the growth and flowering of five common heathland species and on the heathland floristics.

Chapter 5 covers a survey of the boundaries between heath and cleared vegetation. The survey investigates the edges between heathland and cleared areas (such as pastures and gardens) in different locations along the north and east coasts of the state and on Flinders Island. This chapter analyses which environmental, management and disturbance factors influence the condition of coastal heath remnants in Tasmania. In particular, the integrity and health of the heath, with respect to weed invasion and establishment, is linked to different management regimes and types of disturbance at the boundary. It is in this chapter that the presence and impact of edge effects is dealt with most explicitly.

Chapters 2–5 are written as self-contained entities. Therefore there is some unavoidable repetition of review material.

The final chapter, chapter 6, synthesises the results from the previous four chapters. The different responses of heathland from the field experiments are compared and integrated with the findings of the heath–pasture boundaries survey. The resultant picture of heathland responses to a range of edge effects is examined with respect to the research questions and the null hypotheses. The implications for the conservation of fragmented patches of heathland are discussed.

2. Effects of sewage irrigation

INTRODUCTION

Heathlands have long been known to grow on soils with a low nutrient capital (Specht and Rayson 1957; Grundon 1972; Groves 1981a), and many researchers have investigated the effects of adding fertilisers to these systems (Specht 1963; Heddle and Specht 1975; Specht *et al.* 1977; Witkowski *et al.* 1990a; Witkowski *et al.* 1990b). However there is little research into the addition of nutrients in water, particularly in wastewater.

Thomas *et al.* (1997) suggests that wastewater discharge in Australia will increase by nearly 50% between 1994 and 2020, and direct re-use will increase from 1% to 2.5% of the total discharge volume over the same period. Consequently, interest in re-using sewage effluent for irrigating crops or trees has been growing in Australia in the last few years (Lehane 1995). Research has focussed on the positive and negative impacts of re-using sewage effluent in tree plantations (Cromer *et al.* 1982; Stewart and Flinn 1984; Boardman 1991; Myers 1992; Dutch and Wolstenholme 1994; Lehane 1995) and recreational areas (Feigin *et al.* 1991; Cook *et al.* 1994).

In Tasmania, wastewater re-use is limited due to the adequate rainfall in most areas and consequent lack of demand for water (Thomas *et al.* 1997). The Guidelines for the Re-use of Wastewater in Tasmania (Department of Environment and Land Management 1994) also stress that effluent re-use is not to be viewed as 'disposal' but as the use of a secondary resource. In 1994, sewage effluent was being used to irrigate golf courses and municipal gardens in Riverside, and other local councils planned wastewater re-use in the near future (Thomas *et al.* 1997). In addition to the site irrigated in the present study, at least three other areas are currently irrigated or proposed to be irrigated on Tasmania's east coast (R. Mason, Environmental Officer, Break O'Day Council, *pers. comm.* 1996).

With this increase in effluent re-use comes a potential increase in excess water, nutrient and salt problems in the vegetation, soil and groundwater. These potential problems are usually carefully monitored by measuring the water balance (Myers 1992; Myers *et al.* 1996) or nutrients (Stewart and Flinn 1984; Falkiner and Polglase 1997) of the system being irrigated. The context of the present study is to investigate any possible effects on the vegetation of *ad hoc* sewage irrigation of a native heathland. The aim of the study is to monitor any changes in the plant species composition, the amount and proportion of different ground covers and to investigate whether common species react to the irrigation.

THE STUDY AREA

Stieglitz is a small coastal township situated on the St Helens Point peninsula, 8 km east of St Helens on the east coast of Tasmania. Tasmania in general has a temperate maritime climate, which is subhumid and warm on the east coast. The average annual rainfall at St Helens is 780 mm; average maximum temperatures for the summer months are 21.0–22.9 °C and average minimum temperatures for the winter months are 2.5–3.7 °C (Bureau of Meteorology 1999). Figure 2.1 shows the average rainfall by season over the study period, including the long-term average.

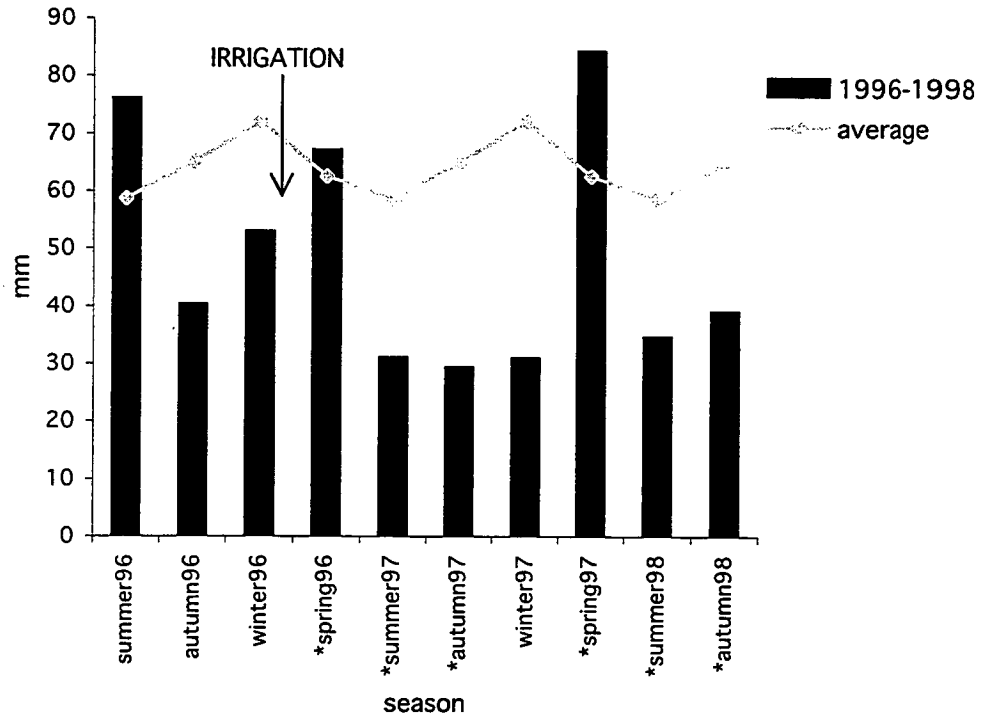


Figure 2.1. Rainfall at St Helens by season. Summer: Dec–Jan–Feb, autumn: Mar–April–May, winter: June–July–Aug and spring: Sep–Oct–Nov. # denotes measurement season

The sewage treatment plant is 1 km from Stieglitz and is surrounded by approximately 50 ha of freehold land owned by the Break O'Day Council. The soils are podzols derived from granite with an iron-cemented gravel and sand 'coffee rock' layer approximately 1–2 m below the surface (Sloane and Weldon 1997). The sewage ponds are on the northern end of a small thin plateau running roughly north–south which has an old coastal escarpment along its eastern edge. The area drains east and northwest into two lagoons, Windmill Lagoon to the east and Chimneys Lagoon to the northwest (Figure 2.2).

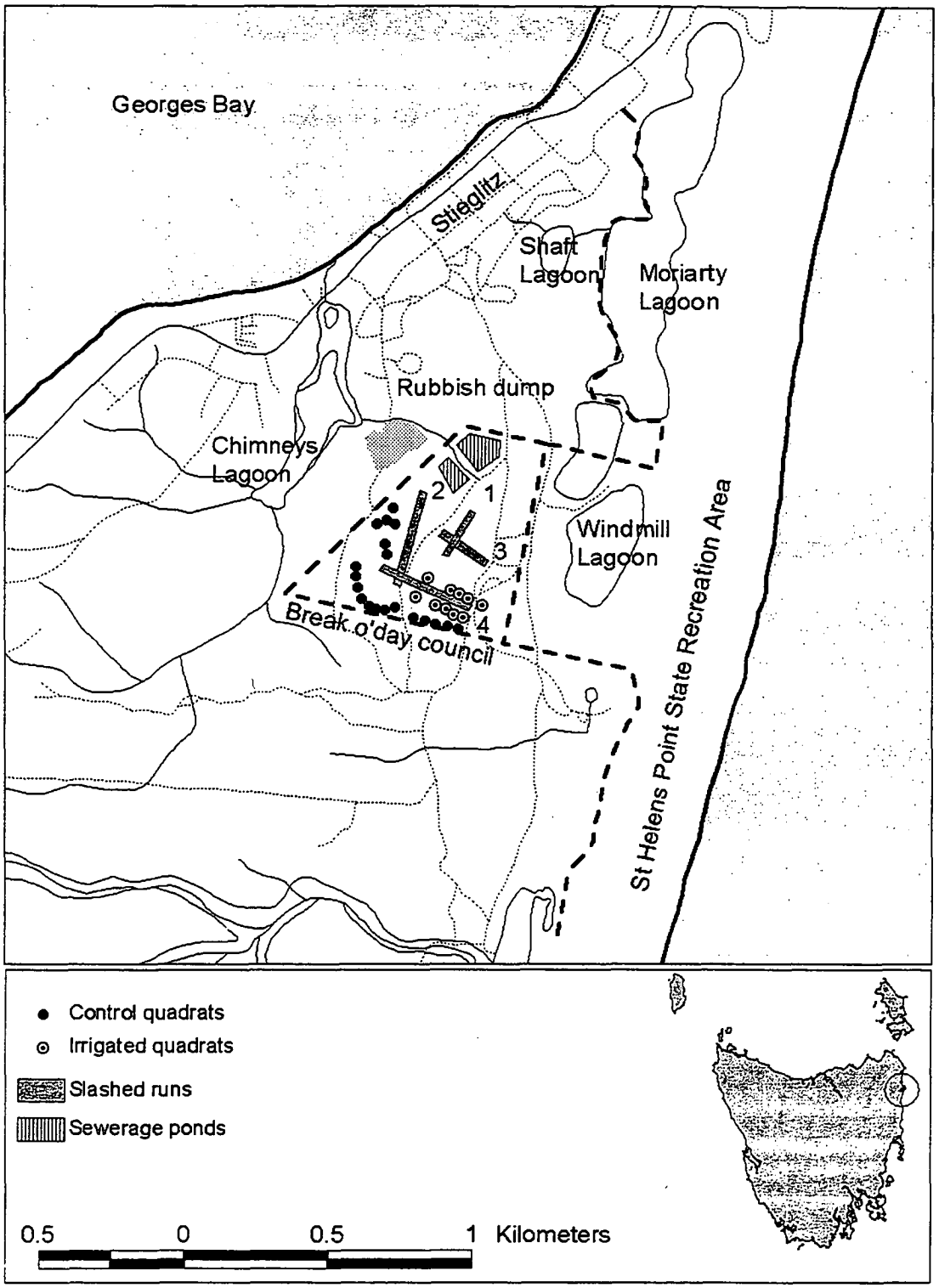


Figure 2.2. Map of Steiglitz area showing study site with sewage ponds, slashed runways, quadrats and surrounding lagoons

The vegetation surrounding the ponds is a heathy low open woodland (Figure 2.3a). The sparse overstorey consists of *Eucalyptus amygdalina* (Black Peppermint), *Allocasuarina littoralis* (Black She-oak) and some *Banksia marginata* (Honeysuckle or Silver Banksia). The most common shrubs are *Leptospermum scoparium* (Manuka or Teatree), *Epacris impressa* (Common Heath), *Leucopogon collinus* (White Beard-heath), *Acacia terminalis* (Sunshine Wattle) and *Boronia pilosa* (Hairy Boronia). Common understorey species are *Hypolaena fastigiata* (Tassel Rope-rush), *Dampiera stricta* (Blue Dampiera), *Lepidosperma concavum* (Sand-hill Sword Sedge), *Schoenus lepidosperma* (Slender Bog-rush) and *Patersonia fragilis* (Short Purple Flag Iris). There were no exotic species found. Ground cover in some areas is sparse. The most recent fire was around 7–10 years before the beginning of the study.

The sewage treatment plant

The treatment plant was built in 1991 and was designed as a facultative two-pond system that stores the effluent and reduces the quantity of water and nutrients. Water is lost through evaporation at an average rate of approximately 11 mm/week. Facultative ponds use both aerobic and anaerobic bacteria to reduce organic matter and nutrient levels. The theoretical detention times are 140 days in the primary pond and 65 days in secondary pond, although the actual times depend on the amount of effluent loading, and stratification and currents within the ponds (Sinclaire Knight Mertz 1996).

Effluent irrigation was not originally planned by the local council. However, the 1995–96 summer was much wetter than usual (Figure 2.1) and the ponds threatened to overflow (R. Mason, Environmental Officer, Break O'Day Council, *pers. comm.* 1996). The council decided to irrigate the surrounding native vegetation as a short-term measure until a better-designed system could be put in place. The council also spread tree and grass seeds over the irrigated area in an attempt to ameliorate the increased nutrient budget. The long-term plan is to pipe the effluent to a holding pond near the local aerodrome (approximately 2 km to the southwest) and irrigate a crop of lucerne or pasture alongside the existing runway strip.

METHODS

Sewage effluent irrigation

Before any irrigation could commence, the heathland vegetation was slashed by the council to allow a clear path for the travelling irrigator. Two parallel lanes 200 and 300 m long and running across the contours (runways 1 and 2) were slashed first (Figure 2.2),

Figure 2.3. a) Heathy woodland at study site showing *Acacia terminals* and *Epacris impressa* flowering under an overstorey of *Eucalyptus amygdalina* and *Allocasuarina littoralis*

Figure 2.3. b) Heathy woodland at study site showing runway slashed for use by travelling irrigator

followed by two more lanes (runways 3 and 4), also 200 and 300 m long, perpendicular to the first two (Figure 2.3b).

The irrigator has a 35 m radius, with large orifice, low pressure, gun-type sprinklers which minimise misting (Sinclaire Knight Mertz 1996). The water for irrigating was taken from the secondary pond outlet near the surface, and was pumped at a fairly constant 53 kilolitres per hour. The total amount of phosphorus and nitrogen spread over the study period is shown in Table 2.1. Averages and standard deviations were calculated from monthly measurements during the irrigation period.

Table 2.1. Total amount of P and N spread by effluent irrigation over study period. Total amount of water for runway 3 was 949 kl and for runway 4 was 2173 kl

nutrient	mean (st dev)	amount per ha
RUNWAY 3 (1.4 ha)		
total nitrogen (kg)	7.57 (1.1)	5.4
ammonium (kg)	0.31 (0.4)	0.2
nitrite (g)	41.15 (46.8)	29.4
nitrate (g)	36.32 (22.7)	25.9
total phosphorus (kg)	2.59 (0.2	1.9
RUNWAY 4 (2.1 ha)		
total nitrogen (kg)	17.33 (2.6)	8.3
ammonium (kg)	0.71 (1.0)	0.3
nitrite (g)	94.16 (107.2)	44.8
nitrate (g)	83.12 (52.0)	39.6
total phosphorus (kg)	5.92 (0.5)	2.8

Table 2.2 shows the schedule of irrigation times and quadrat measurements for the period August 1996 to May 1998. All irrigation amounts and timings were recorded in a log book except for the first and last times. The first few irrigations occurred on runways 1 and 2 before August 1996 and no records were kept of the timing or amount of irrigation water. These two runways were discarded after the early irrigation due to the problem of effluent runoff down the runways back into the sewage ponds. There was possibly one irrigation of runway 3 or 4 in December 1996. This could not be verified because the irrigation logbook was either lost or destroyed in a fire. The emergency nature of the initial irrigation meant that it had occurred before any true baseline measurements were taken. This methodological problem is dealt with below.

Seed planting and soil sampling

The tree seed used by the council was *Eucalyptus globulus* (Tasmanian Blue Gum), collected in 1993 from 15 km south of St Helens at Scamander. *E. globulus* does not occur naturally in coastal heathlands. The seed was sown every 3 m in lines parallel to runways 3 and 4 and

lightly cultivated with a hoe. A total of approximately 2 kg of seed was used. The sowing took place on 17 September 1996.

Table 2.2. Irrigation times, seed plantings and quadrat measurement for Stieglitz sewage irrigation

Date	Irrigation and seed planting	Quadrat measurements
prior to Aug 96	Runways 1 & 2	
Aug 96	Runways 3 & 4	
Sep 96		species–area curve
mid Sep	tree and grass seed planted	
late Sep 96	Runways 3 & 4	
Oct 96	Runways 3 & 4 twice each	
Oct – Nov 96		Spring 1
Nov 96	Runway 4	
Dec 96	possible irrigation	
Feb 97		Summer 1
April 97		Autumn 1
Sep 97		Spring 2
Feb 98		Summer 2
May 98		Autumn 2

The other seed sown was *Secale cereale* (Ryecorn) an exotic grass used to stabilise road cuttings. Ryecorn is the sterile cross between Ryegrass and Wheat. It was purchased from an agricultural supplier and approximately 50 kg, mixed with some *E. globulus* seed, was broadcast by tractor over runways 3 and 4 on 24 September 1996.

Soil samples were collected by the council at the beginning of the irrigation in early August 1996. Irrigating had already occurred on runways 1 and 2, but the samples were taken from around runway 4 which had not yet been irrigated. Around 30 samples were taken using the top 5 cm of soil after clearing the surface humus and litter layer, and then bulked. Final soil samples were collected for the present study at the end of the measurement period in April 1999. The samples were collected from around runway 4, using the top 5 cm of soil as collected previously. Thirty samples were taken and then bulked from the irrigated area, and 30 samples (bulked) from outside the irrigated area.

The soils were air dried, ground and sieved and then analysed for pH and conductivity using a 1:5 soil/water suspension, and for available phosphorus and potassium using Colwell's bicarbonate-extractable phosphorus method. Zinc, manganese, iron and copper were measured using a DTPA acid extract, and nitrates analysed using 2 molar KCl and steam distillation (Raymont and Higginson 1992).

Sampling design and measurements

Data for three species–area curves was collected by identifying all the species in a 50 cm square, then doubling the size and counting the number of new species, and continuing in this

way until no new species were counted for the last two increases (Mueller-Dombois and Ellenberg 1974). The curve flattened out at 25 m^2, so the quadrats used were 5 m x 5 m squares.

Ten quadrats were put 10–20 m from the edge of runway 4, within the radius of the irrigation spray. Each quadrat was placed roughly in a line parallel to the slashed runway using a random distance from the previous quadrat, with the proviso that clumps of *Eucalyptus amygdalina* trees were avoided.

Twenty control quadrats were randomly scattered around runways 3 and 4 at least 15 m outside the edge of the radius of irrigation. Ten of these quadrats were matched with the irrigated quadrats using Bray-Curtis dissimilarity scores based on cover abundance values. One matched control quadrat was destroyed between May and August 1997 by the council laying a pipeline to pump the sewage effluent to the aerodrome. It was replaced with one of the unmatched control quadrats and paired using the original dissimilarity scores.

The 10 matched pairs of quadrats were visited six times over one and a half years (Table 2.2). Slope and aspect were recorded for all 30 quadrats in the first visit. For all visits, identifiable vascular plant taxa were recorded and given a cover class (1, <1%; 2, 1–5%; 3, 6–25%; 4, 26–50%; 5, 51–75%; 6, > 75%) (Mueller-Dombois and Ellenberg 1974). The species that were dominant and subdominant for the overstorey and the understorey were noted. A species was considered dominant if had a cover of greater than 50%, and subdominant if it had a cover of at least 50% that of the dominant species. A 20 m line intercept (the perimeter of the quadrat) was used to measure the percentage cover of live plants, dead plants and bare ground. Species nomenclature follows that of Buchanan (1999). For a full species list see Appendix 1.

Data analysis

Global non-metric multidimensional scaling (GNMDS) was used to investigate whether there were any underlying phytosociological patterns in the first measurement period, and thus assess the adequacy of using it as baseline data. GNMDS is usually more robust to non-linear relationships and variations in the sampling pattern than other forms of ordination (Minchin 1987). Ordinations were calculated using mid-points of cover abundance values and were based on one to four dimensions with 10 random starting configurations. A stress value of 0.2 or less was used as the cut-off for a useable ordination when choosing the number of dimensions. The Multi-Response Permutation Procedure (MRPP) was also used to test for patterns in the phytosociological data, using the software PC–ORD (version 4). The MRPP tests for multivariate difference between treatment groups. This procedure is a non-parametric version of discriminant analysis, and provides the statistics A (within-

group homogeneity) and p (the probability that the observed within-group homogeneity is smaller or equal to that expected by chance) (McCune and Mefford 1999).

Bray-Curtis dissimilarity scores for the paired irrigated and control quadrats were calculated for each time and then plotted to investigate changes in quadrat similarity. The larger the dissimilarity score, the greater the difference between the irrigated and control quadrat, and the smaller the score, the more similar the quadrats.

The paired Students *t*-test was applied to test for significant differences between the irrigated and control treatments for five quadrat variables: species richness, abundance (the sum of the midpoints of all cover values for the entire quadrat), and percentage dead, bare and live cover. The data were tested for significant differences for each measurement season separately. Changes over time within season were also investigated by subtracting the value of a variable in the first season from the same variable in the second season (Spring 2 – Spring 1, Summer 2 – Summer 1 and Autumn 2 – Autumn 1).

The change in the cover of individual plant species over time was also tested for differences between irrigated and control treatments, using the mid-points of the cover abundance values. In this case, the change was calculated over adjacent seasons, i.e. Spring 1 to Summer 1, Summer 1 to Autumn 1 etc. Again, the variable of the second season was subtracted from the same variable in the first season. Species that occurred in less than five quadrats were not used. Change over time was used rather than the absolute cover values because these differed greatly between treatments.

The effect of irrigation on lifeform groups and individual plants was tested using the Chi-squared test on cover changes over time, species dominance in different strata, and presence or absence of different lifeforms in the different treatments. The Yates correction for continuity was used where d.f. = 1. Cases where expected values where less than 5 (for d.f. = 1) or less than 4 (for d.f. = 2) were excluded (Zar 1996).

The analyses were done using the statistical package MINITAB (Release 12.2). Where transformation was needed, either natural log, square root, arcsine or the lambda value suggested by a Box-Cox transformation was used, depending on the type and shape of the data (Sokal and Rohlf 1995). The Box-Cox Transformation estimated a lambda value that minimised the standard deviation of the transformed variable. Different variables and different scores therefore sometimes had different lambda values. The four quadrat variables, overstorey dominant and subdominant species cover and understorey dominant and subdominant species cover, could not be transformed to normality, and the Mann-Whitney *U*-test was used to test for differences in median ranks. The change in individual plant covers over time could also not be transformed to normality and for these data the Kruskal-Wallis rank order test for medians was used as a non-parametric alternative to the Student *t*-test.

RESULTS

Floristic patterns

Quadrat similarity

At the beginning of the study, irrigated and control quadrats were paired using 70% or greater similarity (except one pair which could only be matched to 64%). Because the irrigation occurred in a discrete area along the slashed runways, the two treatments (irrigated and control) could not be randomly allocated, leading to the potential problem of site effects. The aim of pairing the quadrats was to mitigate this problem by ensuring a starting point of high similarity between irrigated and control quadrats, and following these pairs for differences over time.

Six pairs were more similar at the end of the experiment than at the beginning, two pairs were less similar and two pairs were much the same (Figure 2.4). There is no clear pattern of either increasing or decreasing similarity of all quadrat pairs. As a whole, there appears to be greater variation in similarity scores for all quadrat pairs in autumn and spring and much less variation in the similarity of quadrat pairs in the first summer.

Species abundance ordination

The MDS ordination resulted in a three-dimensional solution based on a minimum stress of 0.1409. The ordination shows some floristic differences between the two treatments for the first measurement period. When the first two axes are graphed, the irrigated quadrats are clustered more tightly and the control quadrats are more spread out, despite the cloud of points overlapping (Figure 2.5). There were no obvious differences in the grouping of irrigated and control quadrats along the second or third axes, therefore they are not shown.

Slope and aspect vectors were not significantly correlated with the ordination. Vectors for three variables, percentage bare cover, live cover and total cover abundance, did have a significant correlation with the ordination, and were plotted along with a significant vector for treatment. The bare and live cover vectors are virtually parallel and are therefore closely related, but they are orthogonal to and thus independent of the direction of floristic change.

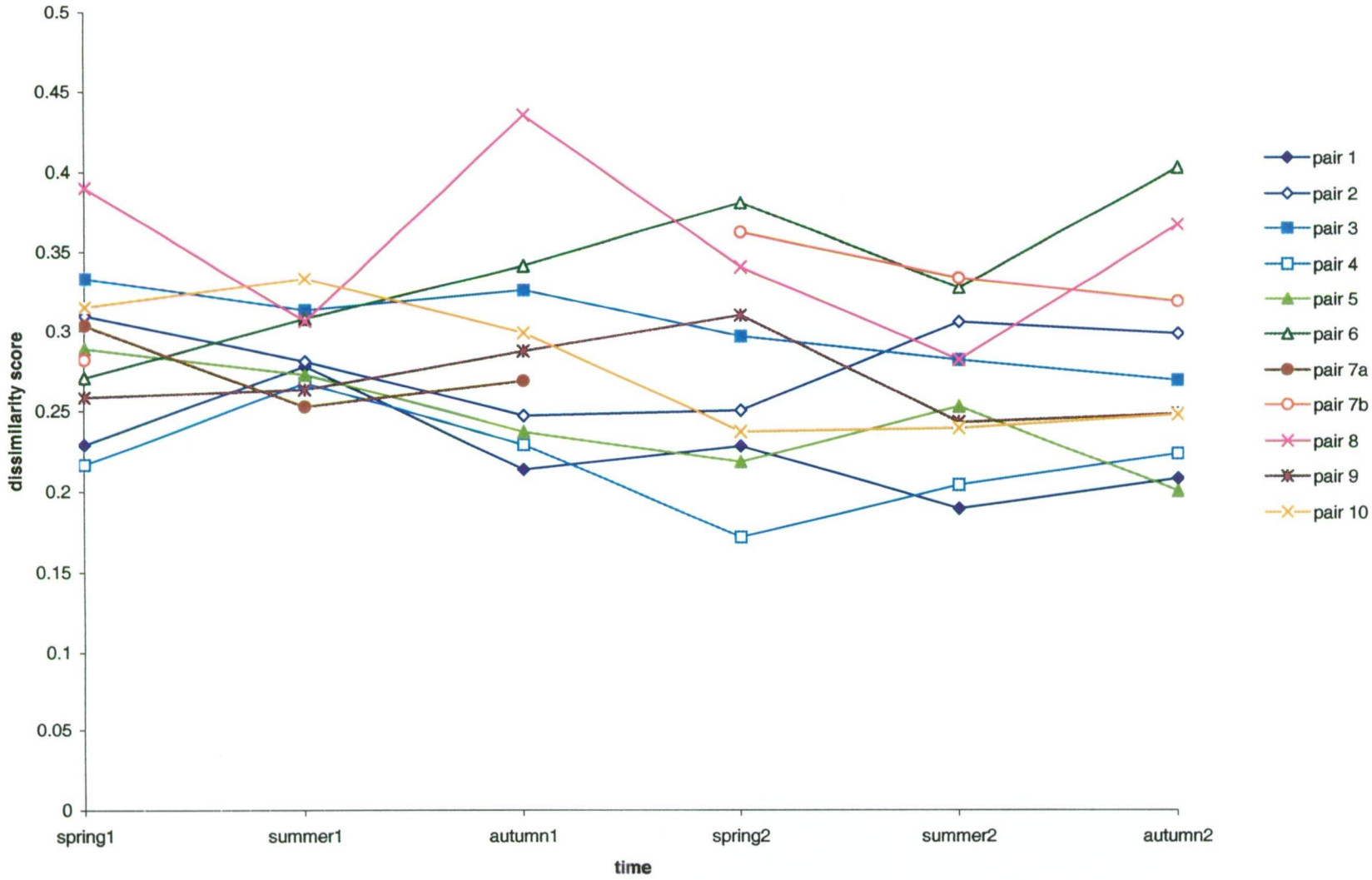


Figure 2.4. Dissimilarity scores between paired irrigated and control quadrats over time, using cover abundances

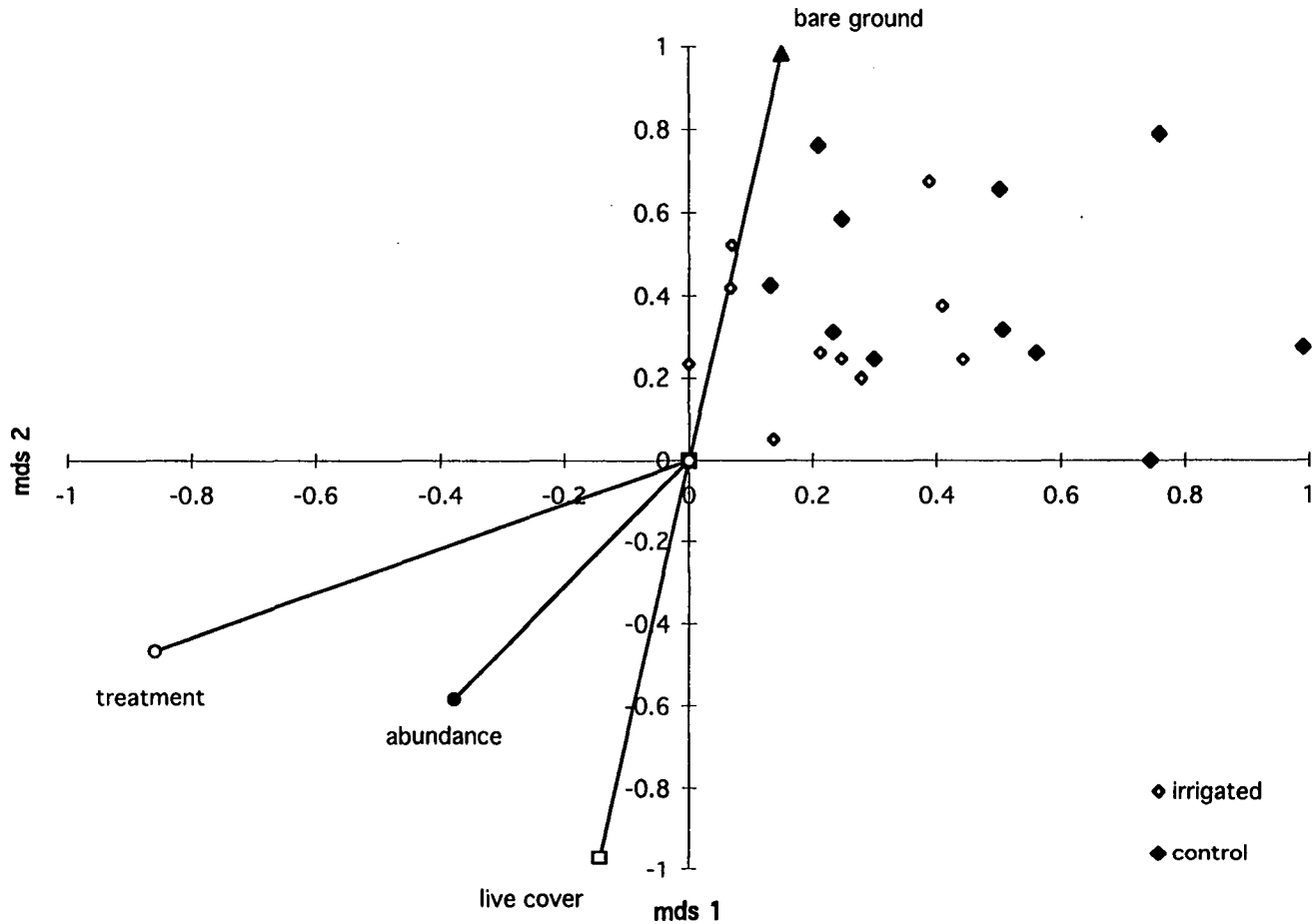


Figure 2.5. GNMDS axes 1 x 2, with significant vectors

The treatment vector scores generated for each quadrat were tested for significant differences between irrigated and control quadrats using the Students *t*-test. The results were significant (P = 0.0086), meaning that there was a relationship between the treatment vector and the floristic gradient in the ordination. Because of this relationship, the treatment vector could be used as a surrogate for the original floristic gradient. The measured variables could then be tested against the treatment vector scores to test for relationships with the original floristic gradient. In the first time period, there was a significant correlation between abundance and the treatment vector scores (P = 0.010). Significant differences in abundance between irrigated and control quadrats therefore do not necessarily indicate a treatment effect — changes in abundance could be due to the initial difference in floristics.

The MRPP results (using cover abundance scores) also showed a significant difference between the irrigated and control groups in the first measurement season (Table 2.3). The closer A is to 1, the more similar the quadrats within each group and the more different the groups are to each other. A negative A value indicates there is less homogeneity within groups than expected by chance (McCune and Mefford 1999). Irrigated and control groups were significantly different in Summer 1, Spring 2 and Autumn 2, but there was no significant difference in Autumn 1 or Summer 2.

Table 2.3. A and p scores for Multiple-Response Permutation Procedure

season	A	p	significance
Spring 1	0.0230	0.0261	*
Summer 1	0.0337	0.0068	**
Autumn 1	0.0090	0.2010	ns
Spring 2	0.0304	0.0153	*
Summer 2	-0.0039	0.5920	ns
Autumn 2	0.0309	0.0208	*

Effects of irrigation on species richness and cover

Species richness, bare ground, and dead and live cover were tested for differences between irrigated and control quadrat pairs for each of the six measurement seasons. The environmental variables (slope and aspect) were tested in the first season and were not significantly different between the irrigated and control quadrats.

There were no significant differences between any variable in response to the treatment in the first 'baseline' season (Spring 1). There was significantly more bare ground and less live cover in the control quadrats in the last season (Autumn 2) (Figure 2.6b, c). Bare ground was always higher in the control quadrats. Dead cover and bare ground increased over the

study period (Figure 2.6a, c) and live cover decreased (Figure 2.6c). There were no significant differences in species richness or in the percentage of dead plant cover in any season. Species richness was highest in Spring 2 and lowest in Summer 1 (Figure 2.6d).

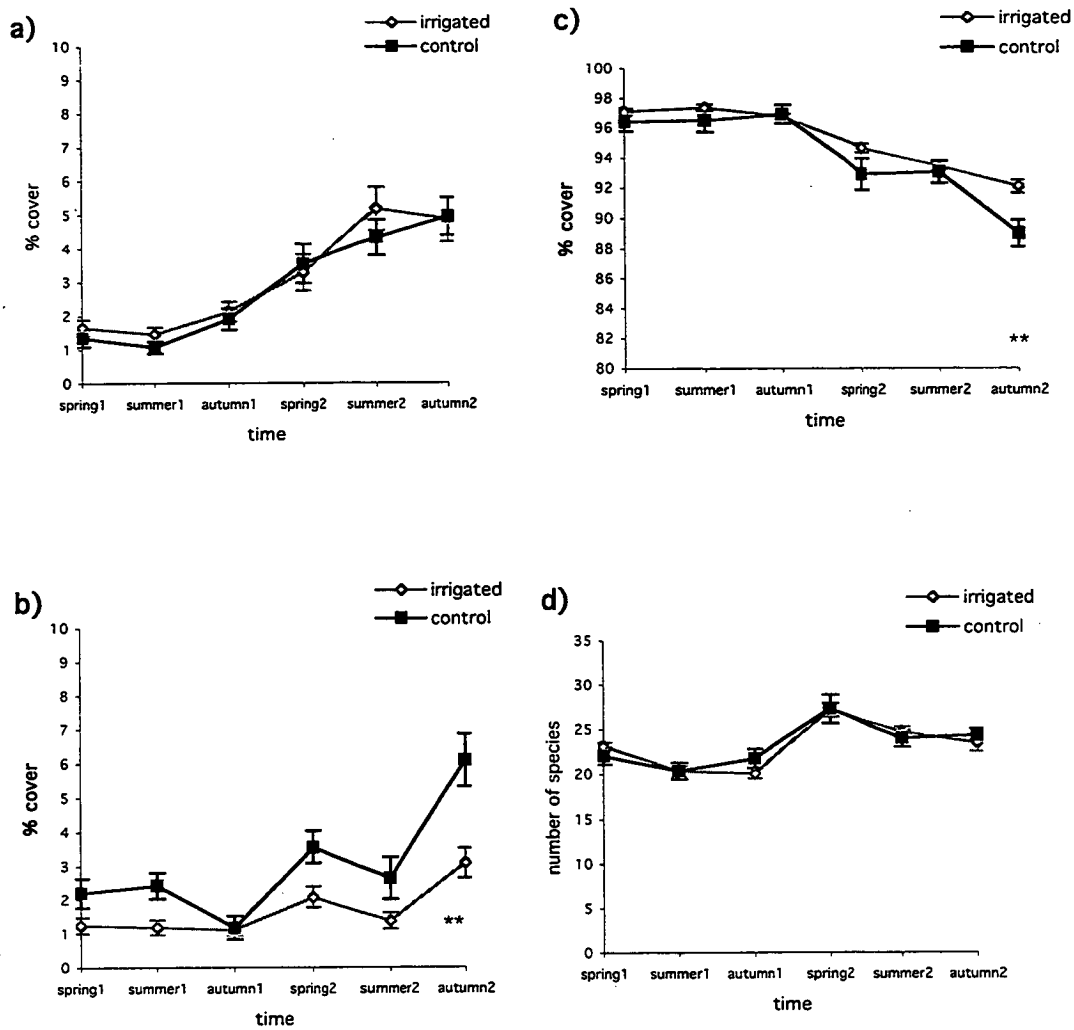


Figure 2.6. Percentage a) dead cover, b) bare ground, c) live cover and d) species richness; averaged for all irrigated and control quadrats for each season. Standard error bars are shown. *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

For change in cover over time, there was a greater decrease in live plant cover in the control quadrats compared with the irrigated quadrats between Spring 1 and Spring 2 (Table 2.4). Between Autumn 1 and Autumn 2 the percentage of bare ground increased more in the control compared with irrigated quadrats, and the percentage of live cover decreased more in the control quadrats.

The three variables overstorey dominant cover, overstorey subdominant cover and understorey dominant cover showed no significant differences between irrigated and control quadrats. The understorey subdominant cover in Summer 1 was higher in the irrigated quadrats than the control quadrats, at $\alpha = 0.05$.

Table 2.4. Average values for dead cover, bare ground and live cover, with standard error in brackets and significance values. Negative values indicate a decrease over time, positive values an increase

	irrigated	control	P	significance
% dead cover				
Spring 2–Spring 1	1.65 (0.39)	2.26 (0.37)	0.50	ns
Summer 2–Summer 1	3.72 (0.55)	3.64 (0.49)	0.197	ns
Autumn 2– Autumn 1	2.72 (0.58)	3.55 (0.54)	0.215	ns
% bare ground				
Spring 2–Spring 1	0.83 (0.33)	1.50 (0.50)	0.698	ns
Summer 2–Summer 1	0.20 (0.22)	0.27 (0.31)	0.854	ns
Autumn 2– Autumn 1	1.97 (0.40)	4.65 (0.58)	0.002	**
% live cover				
Spring 2–Spring 1	-2.48 (0.31)	-3.76 (0.51)	0.027	*
Summer 2–Summer 1	-3.92 (0.45)	-3.91 (0.58)	0.915	ns
Autumn 2– Autumn 1	-4.68 (0.51)	-8.20 (0.70)	0.001	**

ns, not significant; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

Effects of irrigation on individual plants and lifeforms

There were significant differences in cover changes between control and irrigated treatments for three species (Figure 2.7). The cover of *Boronia pilosa* remained the same from Summer 1 to Autumn 1 in the control quadrats, but decreased significantly in the irrigated quadrats (Figure 2.7a, d). The same occurred for *Gonocarpus tetragynus* from Summer 1 to Autumn 1. However, from Autumn 1 to Spring 2, cover increased slightly in the control quadrats but increased significantly more in the irrigated quadrats (Figure 2.7b, e). For *Lepidosperma concavum*, cover remained the same from Spring 1 to Summer 1 for the control quadrats, but increased significantly for the irrigated quadrats (Figure 2.7c, f).

The Chi-squared test was used to test for difference in the number of quadrats in which a species was present or absent, between irrigated and control quadrats. If any cell in the contingency table was zero the species was not used. There were no significant differences for any species in any season. The species were then grouped into seven lifeform groups, namely: tree (woody dicotyledon > 2 m), shrub (woody dicotyledon), graminoid (monocotyledon other than Poaceae or Orchidaceae), grass (Poaceae), herb (non-woody dicotyledon), orchid (Orchidaceae) and fern (Pteridophyta). The same test was used to look for differences in the number of quadrats in which a particular lifeform group was present or absent. There was no significant difference between treatment and control in any lifeform group for any season.

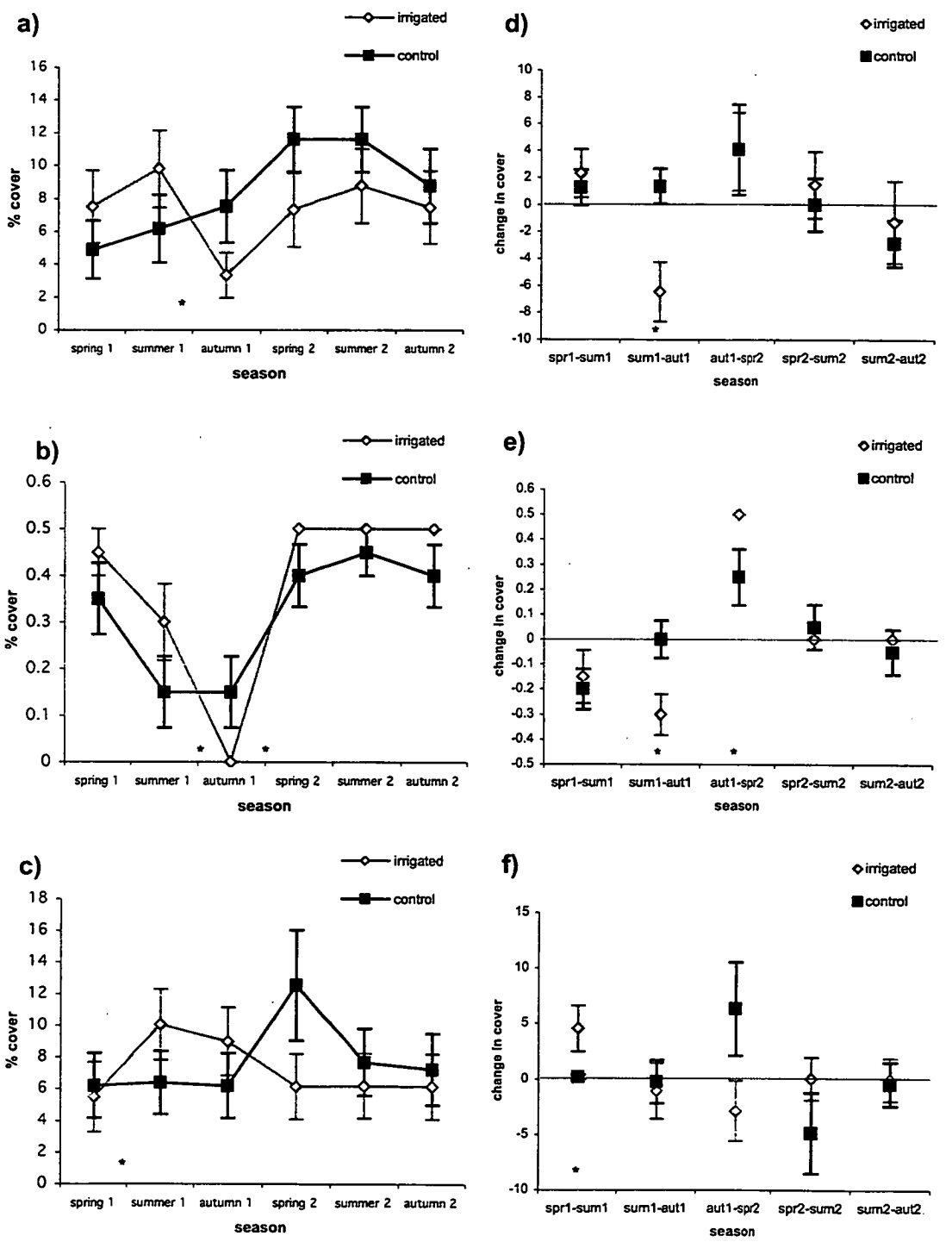


Figure 2.7. Average cover over time for a) *Boronia pilosa*, b) *Gonocarpus tetragynus*, c) *Lepidosperma concavum*. Average change in cover from season to season for d) *Boronia pilosa*, e) *Gonocarpus tetragynus*, f) *Lepidosperma concavum*. Standard error bars are shown. Horizontal lines indicate no change in cover, above is an increase in cover, below a decrease. *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

Seeds and soil

No *Eucalyptus globulus* or *Secale cereale* seedlings were found. The runways and areas between quadrats were searched as well as the quadrats because it could not be verified whether any seeds had been planted within the quadrats. There were no exotic species of any type found during the entire period of the study.

Table 2.5 shows the soil analysis results for prior and post irrigation. Phosphorus, copper and manganese appear to have risen in both irrigated and control quadrats after irrigation, and nitrate and zinc appear to have decreased. Potassium has a lower value in the control quadrats after irrigation but a higher value in the irrigated quadrats. There is an unusually high value for iron prior to irrigation.

Table 2.5. Soil nutrient analyses before and after irrigation

sample	pH	cond dS/m	P ppm	NO_3 ppm	Cu ppm	Fe ppm	Mn ppm	Zn ppm	K ppm
BEFORE									
non-irrig	4.6	0.04	0.0	33.6	0.04	251.5	2.45	0.52	41.0
AFTER									
control	4.8	0.03	8	< 5	< 0.1	45	4.6	0.3	33
irrigated	4.9	0.05	7	< 5	< 0.1	68	4.1	0.1	79

cond, conductivity

DISCUSSION

The ordination of the first measurement period shows that it has to be used cautiously as baseline data because there is a floristic gradient along which the irrigated and control quadrats were separated. The MRPP analysis confirmed that the irrigated and control quadrats were significantly different groups. This difference appears despite the fact that quadrats could be paired to a high level of phytosociological similarity (usually greater than 70%). Meaningful results were extracted from this original floristic pattern by testing for correlation between a variable and the treatment vector scores, which were used as a surrogate for the floristic gradient. Total cover abundance was significantly correlated with the treatment vector, and was therefore discarded from further analysis and discussion.

Apart from total cover abundance, the other variables show that there were no significant differences between the two treatments during the first measurement period. There were no differences in the environmental site variables, slope and aspect. Species richness, bare ground, dead and live cover, and the cover of over- and understorey dominant and subdominant species were not significantly different at the start of the experiment. This establishes that the first measurement period can still be used as a reliable baseline for

variables other than abundance, despite the fact that it occurred after some of the irrigation had already taken place. The baseline therefore allows for plausible comparisons to be made over the period of the study.

Sewage irrigation only had a small effect on the Stieglitz heathland community. In general, all the significant differences show the same trend of slightly increased growth. Although seasonal changes were not the focus of the study, differences between seasons were often more highly significant than differences between the two treatments (data not shown). Increased growth under effluent irrigation is a common response in tree plantations. Hanna (1994) reported high productivity and growth of *Eucalyptus globulus* (Blue Gum) and *E. grandis* (Flooded Gum) after irrigation with secondary treated effluent water. Stewart *et al.* (1990) found an extremely high mean annual biomass increment for *E. grandis* after effluent irrigation in Victoria. Application of sewage sludge and additional copper fertilisation to a conifer plantation on a sandy soil in Denmark showed an increase in tree growth of 30–70% (Olesen and Mark 1991). Increased growth was also the response of *Banksia prionotes* (Acorn Banksia) trees to the incursion of nitrate-polluted groundwater from adjacent agricultural land (Grigg *et al.* 2000). Although the increase in plant growth at Stieglitz is not in the same scale as the growth response reported for trees in plantations and native communities, it does show a comparable result. This response could be caused by the addition of nutrients and/or the addition of water.

Plant growth and nutrient addition

The nutrients and trace elements measured at Stieglitz are within typical values for municipal effluent throughout Australia (Lehane 1995) and the world (Feigin *et al.* 1991). Although the irrigation did not continue over a long time period, the input of nutrients into a low-nutrient system was relatively high when compared with estimated nutrient uptake levels from a *Banksia ornata–Casuarina pusilla–Xanthorrhoea australis* heath in South Australia (Specht 1966). Nitrogen, magnesium and calcium were applied at or slightly above nutrient uptake levels. However the amounts of phosphorus and sodium added were an order of magnitude greater (Table 2.6).

The increase in plant cover at Stieglitz is likely to be due to the addition of nutrients, in particular phosphorus and nitrogen, to soils that had relatively low initial nutrient levels. Adding nutrients in the form of agricultural fertilisers to heath has been shown to result in increased growth (Connor and Wilson 1967; Specht *et al.* 1977). Similarly, the addition of nutrients in sewage water to tree plantations also caused increased growth and productivity (Stewart *et al.* 1990; Hanna *et al.* 1994).

Table 2.6. Nutrients distributed in effluent water in present study (kg/ha) compared with nutrient uptake by heath in South Australia (kg/ha/year)

	Stieglitz, TAS	Dark Island Soak, SA
total nitrogen	7	6.5
total phosphorus	2.43	0.14
sodium	80	1.2
chloride	100	nd
magnesium	3	1.9
calcium	9	5.6

nd, no data

In an experiment where heath was fertilised with extremely high levels of nutrients (1400 and 750 kg/ha for phosphorus and nitrogen, respectively), greatly increased growth of both seedlings and mature heath plants was reported (Specht 1963). When the experiment was revisited nine years later, the increased growth was found to have sped up the life cycle of some species of the heath community and caused them to die many years earlier than usual (Heddle and Specht 1975). The short time period of the present study means it is not possible to know if the slight growth increase is part of the same process.

Plant growth and water use

The increased plant cover at Stieglitz could also be explained by the addition of water during and prior to a relatively dry period. The effluent water was added over winter and spring 1996, which were drier than average and about average, respectively (Figure 2.1). The many drier than average seasons over the study period could also explain the decrease in live cover and increase in both dead cover and bare ground for both treatments over the period of the study.

Other researchers have found that the addition of water adequately explained increased growth following effluent irrigation. In a wastewater irrigation experiment run over four sites in Victoria, Stewart and Flinn (1984) found that the increased growth of *Eucalyptus robusta* (Swamp Mahogany), *E. saligna* (Sydney Blue Gum) and *Casuarina cunninghamiana* (She-oak) was comparable to trees watered with river water alone. Research into the growth response of *Pinus radiata* (Radiata Pine) in Victoria in sandy soils also showed that the annual volume increment was directly related to the addition of water, with phosphate fertilisers (with and without irrigation) having no effect (Cromer *et al.* 1982). However, Grigg *et al.* (2000) measured a 13-fold increase in the biomass of *Banksia prionotes* trees next to agricultural land, due to the presence of both groundwater runoff polluted with nitrates and fertiliser drift. The same could be true of the results at Stieglitz, with both the additional water and the nutrients contributing to the increased heathland plant growth.

In a plantation, least water is used when the trees are seedlings and as the trees grow their water use increases to a peak at canopy closure, after which it remains relatively steady (Lehane 1995). The *E. globulus* and *Secale cereale* seed planting by the Council in the present study would therefore not have assisted in water use in the short-term, even if they had established.

The small positive growth response at Stieglitz is in direct contrast to the results of a similar (unpublished) monitoring experiment at Freycinet, also on the east coast of Tasmania. At this site there was a very high rate of plant death, particularly of *E. tenuiramus* (Silver Peppermint), *Banksia marginata* and *Leptospermum scoparium*, and large areas of bare ground directly after irrigation with sewage effluent. The deaths were probably due to waterlogging, as some areas had wet heath growing on clayey soils with poor drainage. After 4–5 years the heath community had dramatically recovered, with re-seeding of eucalypts and the worst areas colonised by *Lepidosperma concavum* (L. Gilfedder, Vegetation Conservation, Research and Management, Resource Management and Conservation Division, Department of Primary Industries, Water & Environment, *pers. comm.* 2000). The different results at Stieglitz are probably due to the high permeability of the sandy soils and consequent lack of waterlogging.

Individual plant responses and heath species composition

The three species that responded to irrigation did so in different ways, with cover increasing in some cases and decreasing in others. Investigation of changes in species composition based on dissimilarity scores between quadrat pairs did not show any pattern of increasing similarity or difference over the study period. It is not possible to draw any conclusions about unique heath species or community responses from only these two examples. However, the many examples of different tree species showing different responses to effluent irrigation in the literature (Stewart and Flinn 1984; Hopmans *et al.* 1990; Stewart *et al.* 1990; Hanna *et al.* 1994; Myers *et al.* 1996) indicate that a more detailed study would be likely to find comparable differences for heath species.

The lack of exotic species throughout the study is an interesting finding, particularly since other research shows an increase in exotics after the addition of nutrients. Connor and Wilson (1967) found exotic weeds to become established after adding fertilisers, and Heddle and Specht (1975) found an increase in the number of herbaceous species, including many exotics, after fertilisation.

It is notable that there were no major plant deaths or obvious signs of deformation or deficiency in the heath at Stieglitz. Other research shows significant effects from an overload of particular nutrients or an imbalance of micronutrients. In Scotland, sewage sludge produced from domestic wastewater treatment was used as preparation for a spruce

plantation in a heathland. Usually *Calluna vulgaris* (Heather) would be killed by high doses of nitrogen and phosphorus fertilisers and a herbicide treatment prior to planting the spruce seedling, but in an experiment by Dutch and Wolstenholme (1994) the same effect was achieved by using two applications of sewage sludge. Stewart and Flinn (1984) found growth of all eucalypts in Victoria was reduced by iron deficiency, and at another site *E. saligna* had a low survival rate attributed to high salinity. At Wagga Wagga, copper deficiency was thought to cause twisted stems and branches of *E. maculata* (Spotted Gum), and magnesium deficiency to cause yellowing and tip death of *Pinus radiata* (Lehane 1995).

Although the nutrient additions at Stieglitz were relatively high for a low-nutrient heath, the levels were still very small compared to those causing the effects described above. The period of irrigation was also probably much too short to produce any nutrient deficiency effects.

Environmental impacts of irrigation

Despite the fact that the addition of water and nutrients at Stieglitz did not have a detrimental effect on the vegetation, it is likely to have exceeded the holding capacity of the plants and soil, therefore causing runoff and potential downstream impacts. Highly permeable sandy soils and an impermeable layer below a shallow soil layer are generally considered unsuitable for effluent irrigation due to problems with deep drainage or run off (Department of Environment and Land Management 1994; Lehane 1995). According to these criteria, the sandy soils over coffee rock at Stieglitz are quite inappropriate for irrigating. Recharge to the groundwater is likely in this situation, and has occurred due to the removal of heath vegetation on similar sandy soils underlain by an impermeable layer in Western Australia (Grigg *et al.* 2000). However, at Stieglitz, the condition of the groundwater — relatively poor quality, with poor yields and quite saline (Sloane and Weldon 1996) — means the effects may not be too serious.

The three nutrients most likely to cause environmental problems are nitrogen, phosphorus and salt. Nitrate values appear to have decreased in both control and irrigated quadrats after irrigation, but this could be due to the samples being taken from different locations before and after irrigation, or different handling of the samples before analysis. Nitrate is easily taken up by plants but is also the most susceptible to leaching and eutrophication. In a study of large *Banksia prionotes* 'hedges' caused by incursions of nutrients and water from agricultural land, the hedge biomass was found to be enriched with nitrogen (Grigg *et al.* 2000). However, given the relatively small amounts used in the present study, and the uptake rate of the heath, it is unlikely to have caused any off-site problems in the short-term.

Most soils will bind up excess phosphorus, with eventual saturation and subsequent leaching occurring only after many years (Lehane 1995). However, the sandy soils at Stieglitz would have a low phosphorus retention capacity and the heath community a low phosphorus uptake rate, so leaching by soil moisture throughflow along the coffee rock horizon and subsequent increase in the nearby lagoons is likely. The apparent increase in phosphorus values in the control quadrats could be due to spray drift from the effluent irrigation or because the soil samples were not taken from precisely the same locations before and after irrigation.

Salt can build up round the root zone if effluent loading rates are high, but this can be delayed by planting salt-tolerant species and leaching via rainfall (Lehane 1995). Salinity values in the soil at Stieglitz, as measured by conductivity, appeared to have stayed the same over the period of the study. Therefore, despite the low salt uptake rate by heath and the relatively dry weather during the study, salinity does not appear to be a problem.

CONCLUSION

Irrigating a native heathland with sewage effluent is an unusual situation in terms of both ecology and management, as there would not normally be any incentives to irrigate heathland with wastewater. This monitoring experiment took advantage of a crisis situation that was over relatively quickly and that appeared to have no dramatic negative effects in the short term. There was no wholesale death or obvious disease of the heathland plants. The general response was of slightly increased growth, which could have been due to either or both the added nutrients and the added water. The native species composition did not change over the short term and, more significantly, no exotic species were found. Returning to the Stieglitz heath in 10 or 20 years time might produce quite different results, as would a study of the effects of the effluent on the surrounding wetlands. Another productive area for future research would be to design an experiment which included water-only and fertiliser-only treatments over a range of soil types and areas, as well as more accurate measurements of above- and below-ground plant growth.

3. Effects of fertilisers, weed seeds and fire

INTRODUCTION

Heathlands are subject to many different types of disturbance, including clearing, fire, nutrient accession, weed invasion and grazing. The inputs of nutrient accessions in heathland were the subject of several studies in the 1960s and 1970s. Since then, it has been assumed that any input of nutrients to heathlands will be detrimental (e.g. Williams (1991)). Nutrient increases can often result in the invasion of exotic species. Fire is well-known to be an important part of heath ecology, and yet the interactions between fire, nutrients and weed invasion have not been widely investigated. This experiment is designed to look at the interactive effect of fire, fertilisers and weed invasion in heathland in the Peter Murrell Nature Reserve in Tasmania.

Nutrient additions

Previous research into the effects of increased nutrients on heath vegetation have usually involved large amounts of fertiliser. Specht (1963) added fertilisers to a heath at Dark Island near Keith, South Australia, to gradually raise the levels of nitrogen, phosphorus, calcium, magnesium, potassium and sodium in the sandy soil to those found in grassy forest and savanna woodland in other parts of South Australia. Specht *et al.* (1977) used similar high levels of fertiliser on a sand heath on North Stradbroke Island, Queensland, in order to raise the level of nutrients to that of improved pasture. Other experiments have aimed to 'correct' phosphorus, nitrogen and/or trace element deficiencies in the soil (Connor and Wilson 1967; Mendoza *et al.* 1995). Table 3.1 summarises the fertiliser amounts used in various experiments on heathland in Australia.

Although there has been some interest in the effects of small additions of nutrients to heath from such sources as ashbeds (e.g. McMaster *et al.* 1982; Witkowski *et al.* 1990a; Witkowski *et al.* 1990b), there are few experiments addressing the possible effects of other low inputs of nutrients. Examples of such inputs might be drift from aerial top-dressing of improved pasture or forest plantations, erosion from agricultural areas, leaching through sandy soils from disposal sites, runoff from urban areas (including septic tank overflow), litter and excreta from recreational use (for example horse and other animal faeces), rubbish dumping and dust from road- or track-building material. This experiment simulates drift from aerial top-dressing of pasture. One aim of the present study is to find out if such small amounts of nutrients have any effect on heath vegetation.

Table 3.1. Australian research on heath using fertilisers. The figure(s) before the plus sign denotes the establishment amount and the figure after the plus sign the yearly maintenance amount. All amounts in kg/ha except Mo in g/ha. Adapted from Specht *et al.* (1977)

Fert in kg/ha		SUPER	P	S	N	K	Ca	Cu/Zn	Mo	Reference
SE Aust agriculture	Pasture	200 + 100		(super)[1]	legume[2]		(super)	8–16		(Specht *et al.* 1977)
Dark Island, SA	Pasture		19 + 10	20 + 10	legume	—				(Specht 1963) and
	Heath	**240–1940**	10–134	10–140	16–124	—				(Heddle and Specht 1975)
Qld agriculture	Pasture	630 + 250		(super)	legume	125 + 60	(super)	8	250g	(Specht *et al.* 1977)
Beerwah, Qld	Pasture		60 + 24	63 + 25	legume	65 + 32				(Specht *et al.* 1977)
	Heath	**1200** (+rock)	38+195	120	70	210				(Connor and Wilson 1967)
Moreton Is, Qld	Pasture/ heath		56	81	17+ legume	81				(Specht *et al.* 1977)
Stradbroke Is, Qld	Mined		52	6	115	52				(Thatcher and Westman 1975)
	Heath	**825**	95	138	63	52				(Specht *et al.* 1977)
Tasmania agriculture	Pasture		5–10 max 30							Dept Agricultural Science, University of Tasmania
Howden, Tas	Pasture	250 + 100	23 + 9	30 + 12	legume	50 + 25	55 + 22	8.8 Cu	150g	Impact Fertilisers, Tasmania
This research	Heath[3]	**25–50**	2–4.5	3–6	—	mur pot[4]	6.5–11	2.2–4.4	37.5–75g	

[1] present in superphosphate, [2] nitrogen added by sowing legumes, [3] quarter and half general agricultural maintenance amounts, [4] muriate of potash: 100–200 establishment plus 50–100 maintenance

Fire and nutrients

There have been few experiments investigating the interaction between fire and nutrient additions in heathlands. Other than studies into nutrient additions from the ashbed, fire has usually only been accidentally included when researching fertiliser additions to heath, and the timing has varied. For example, the series of experiments in Dark Island were burnt in summer wildfires before or after fertilising, with more uncontrolled fires reburning the sites during the course of the experiment (Specht 1963). The fertilisation experiment by Specht *et al.* (1977) also occurred three years after a fire (season not noted), but otherwise did not include fire. The second aim of the present study is to include fire by controlling the timing of both the burn and the nutrient additions, and to investigate the interactive effects of fire and fertilisers on heathland vegetation.

Weed invasions

Invasion by exotic plants into a variety of native communities is a problem for conservation in Australia and has been widely researched, particularly the importance of disturbance (Amor and Piggin 1977; Fox and Fox 1986; Hobbs and Atkins 1988; MacDonald *et al.* 1988; Hobbs 1989). Weeds have been found to increase after fire (Milberg and Lamont 1995) and due to raised nutrients (Connor and Wilson 1967; Hobbs and Atkins 1988; Cale and Hobbs 1991; Hester and Hobbs 1992) and/or soil disturbance (Hobbs and Atkins 1988).

In heathlands, exotic species have become established following the addition of fertiliser (Connor and Wilson 1967), due to changes in fire regime (Hopper and Muir 1984; MacDonald *et al.* 1988) and as part of the rehabilitation process after sand mining (Thatcher and Westman 1975) but little research has focussed on how weeds respond to the combined effects of burning and fertilising. The third aim of the present study is to investigate the interactions between the addition of fertilisers, the disturbance caused by fire, and weed invasion in heathland.

THE STUDY AREA

The study site is within the Peter Murrell Nature Reserve located on the southeast coast of Tasmania, approximately 15 km south of Hobart (Figure 3.1). The Peter Murrell Nature Reserve covers an area of 133 ha and is surrounded on three sides by the Peter Murrell Conservation Area (135 ha). The Nature Reserve was proclaimed in August 1997 (half-way through the experimental period); prior to this the site was managed by the Department of Community Services and Health, and was designated to be developed for housing.

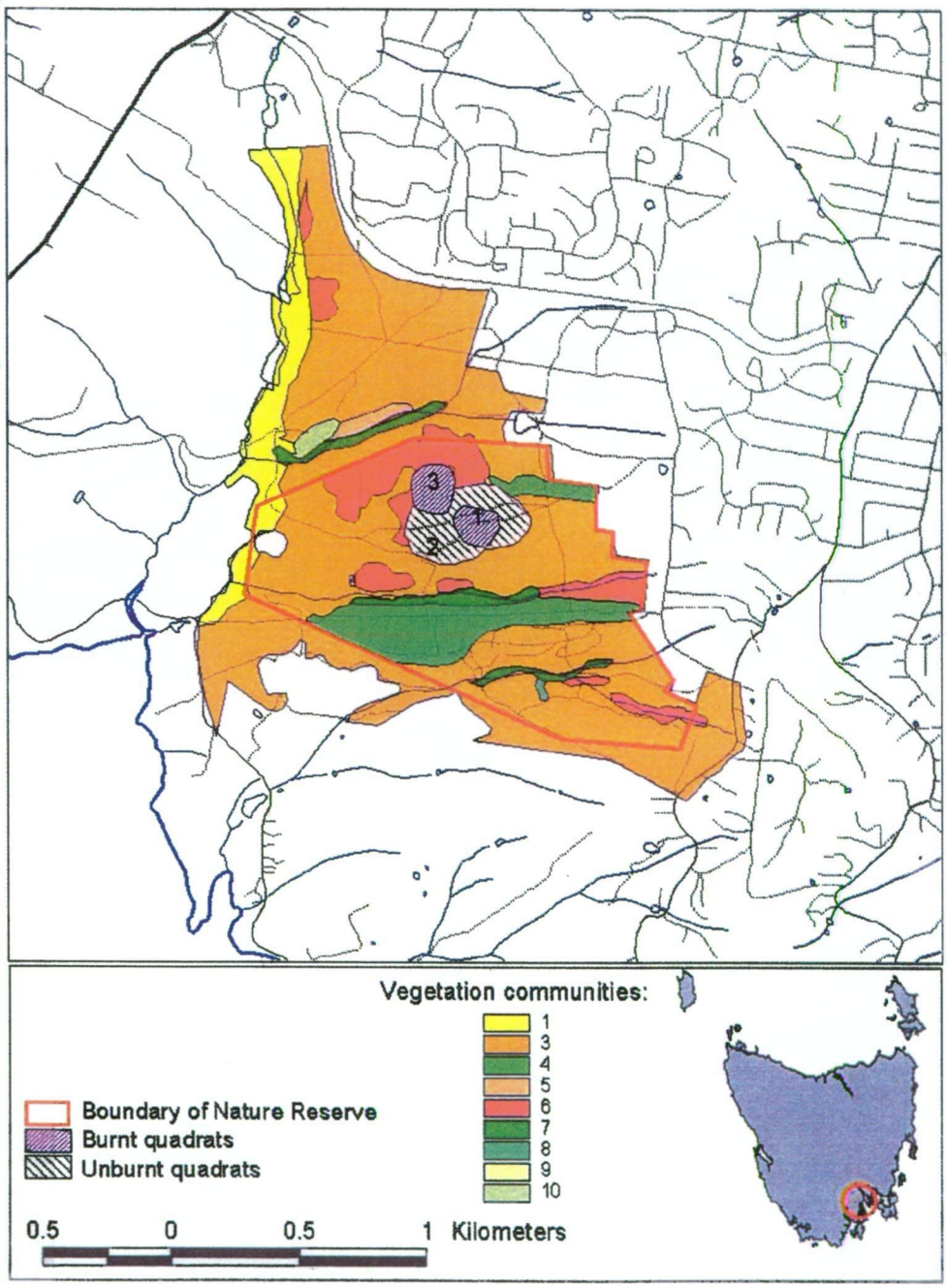


Figure 3.1. Map of Peter Murrell Nature Reserve showing vegetation types, study site, burn blocks, quadrats and surrounding Peter Murrell Conservation Area. Vegetation: **1** *Eucalyptus amygdalina – E. viminalis – E. obliqua* shrubby forest; **2** *Eucalyptus ovata* shrubby forest; **3** *Eucalyptus amygdalina* heathy coastal forest woodland; **4** *Eucalyptus amygdalina* heathy forest on sandstone; **5** open heath; **6** closed heath; **7** sedgeland; **8** grassland; **9** wetland. Modified from Parks and Wildlife Service Tasmania (1997)

Tasmania has a temperate maritime climate, which is subhumid and warm on the east coast. Figure 3.2a shows the average monthly rainfall over the study period for three locations around the study site. Kingston and Blackmans Bay are the closest stations but have short-term records, therefore both the short- and long-term records for Hobart are included. Most of the Peter Murrell Nature Reserve is within the sandstone-based Maranoa Heights land system, and outcrops of metamorphosed sediments occur where this rock type contacts the Ragged Tier tertiary basalt (Duncan and Duncan 1995).

Vegetation

The main vegetation type within the Nature Reserve is *Eucalyptus amygdalina* (Black Peppermint) heathy woodland (Kirkpatrick 1999). The experiment was carried out in areas of closed heath without a eucalypt overstorey within the woodland (Figure 3.3a). The heath has a closed shrub layer up to 80 cm high which is dominated by *Leptospermum scoparium* (Teatree or Manuka) and *Allocasuarina monilifera* (Necklace She-oak). Other common shrubs are *Amperea xiphoclada* (Weaver's Plant or Broom Spurge), *Aotus ericoides* (Golden Pea), *Baeckea ramosissima* (Baeckea or Rosy Heath Myrtle), *Bossiaea cinerea* (Showy Bossiaea), *Epacris impressa* (Common Heath) and *Leucopogon collinus* (White Beard Heath). The ground layer is around 30 cm tall and dominated by graminoids, conspicuously *Hypolaena fastigiata* (Tassel Rope-rush) and *Schoenus lepidosperma* (Slender Bog-rush). *Pteridium esculentum* (Austral Bracken) was ubiquitous in the heath (see Appendix 2 for full species list).

The heath corresponds with the *Eucalyptus amygdalina – Baeckea ramosissima – Tetratheca labillardierei* (Leggy Black-eyed Susan) vegetation community described by Kirkpatrick (1999), which is most similar to the heathy *E. amygdalina* coastal forest Statewide community (Kirkpatrick 1999). Of the Tasmanian heath communities described by Kirkpatrick (1999b), the 'Manuka Dry Heath' best describes the heath at the Peter Murrell Nature Reserve.

Disturbance

The Conservation Area surrounding the Nature Reserve is itself surrounded by suburbs, an industrial area and a pony club. These areas are sources of a range of exotic weed species, as is the previous dumping of household and garden wastes, shells and soil. Most of the weeds in and around the Reserve are not highly invasive, except for *Cytisus* spp. (Broom), *Chrysanthemoides monilifera* (Boneseed), *Cortaderia* spp. (Pampas Grass), *Erica lusitanica* (Spanish Heather) and exotic *Acacia* (Wattle) species (Parks and Wildlife Service Tasmania 1997). *E. lusitanica* was found in the open heath but not in any quadrats, and *Hypochoeris radicata* (Cat's Ear) and *Holcus lanatus* (Yorkshire Fog) were the only

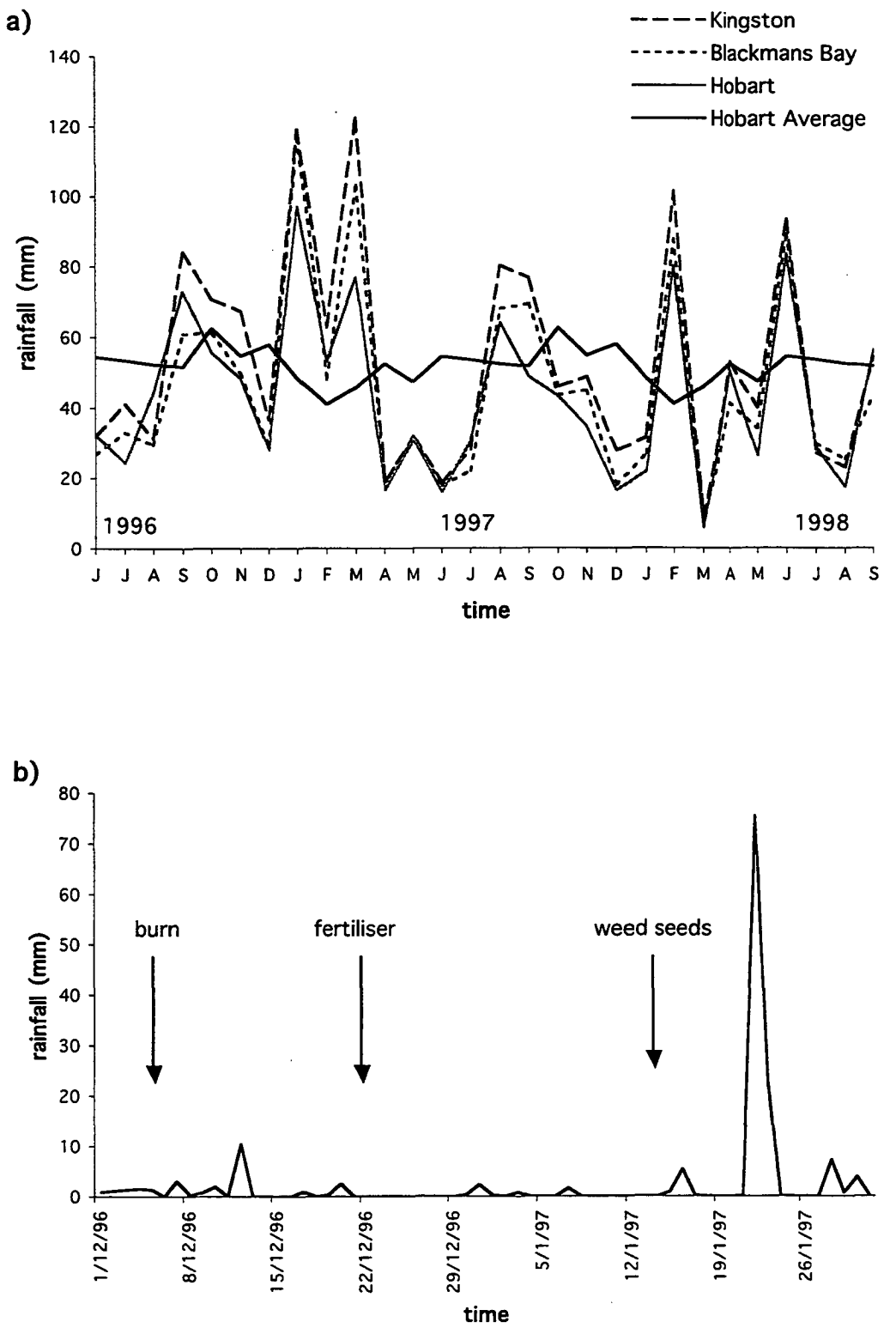


Figure 3.2. a) Monthly rainfall for Kingston, Blackmans Bay and Hobart over the study period compared with the long-term average rainfall for Hobart, b) daily rainfall averaged for Kingston, Blackmans Bay, Bull Bay and Taroona for the two months during which the burn, first fertiliser treatment and weed seeds treatment took place

Figure 3.3. a) Closed heathland at study site showing *Pteridium esculentum* and *Leptospermum scoparium*

Figure 3.3. b) Fire front and fire truck at burn block 3

exotic species found in any quadrats, apart from the exotic grass seeds that were sown as part of the experiment (see below). Other disturbances prior to the area becoming a Nature Reserve were trail-bike riding, horse-riding, people walking dogs, dumping of car bodies, and firewood and rock collection. Some of these are still threats to the Reserve (Parks and Wildlife Service Tasmania 1997).

Fire history

The Nature Reserve would have experienced many fires of different sizes and intensities over the last two decades, but there were no consistent records or maps of the burns. The study site itself was probably burnt in January 1988, based on *Leptospermum scoparium* growth rings, local Fire Brigade records, local knowledge and Ziegeler (1994). Prior to this, the area could have burnt in November 1984, based on local Fire Brigade records, and prior to that in approximately 1976, based on *Banksia marginata* (Honeysuckle or Silver Banksia) node counts. If these estimates are correct, fire frequency in the last 20 years has been relatively high, i.e. a fire every 6–7 years.

Duncan and Duncan (1995) suggest that frequent cool burns will decrease species diversity and can result in the community becoming dominated by *Pteridium esculentum* (Bracken). Bracken increases at the expense of shrubs which do not have time to set seed or regenerate by coppicing or from rootstocks because of the frequent fires. The large amounts of bracken and relatively low species diversity in the closed heath at the Nature Reserve support this suggestion and add weight to the above deduction of frequent firing.

METHODS

The main focus of the experiment was to look at the effect of fertilisers alone, the effect of fire alone and the combined effect of fertilisers and fire. An additional aspect of the experiment was to investigate if there was any response to fertilisers from exotic weeds. The weed response was determined by experimental manipulation (fertilisers and exotic grass seeds added) and background weed invasion (fertilisers only). Due to the size of the quadrats and the limited space in the heathland study site, there was no exotic grass seed without fertiliser combination. However, background weed invasion was possible in both fertilised and unfertilised quadrats.

Therefore, the three treatments were fire, fertilisers and exotic grass seeds, in ten combinations (Table 3.2). The fire treatment was either burnt or unburnt, the fertiliser treatment was no fertiliser, level 1 or level 2 fertiliser and the exotic grass seed treatment was seed or no seed.

Table 3.2. Experimental treatment combinations and abbreviations used in text

code	description	code	description
BC	burnt control	UC	unburnt control
BF1	burnt, fertiliser level 1	UF1	unburnt, fertiliser level 1
BF2	burnt, fertiliser level 2	UF2	unburnt, fertiliser level 2
BSF1	burnt, seed, fertiliser level 1	USF1	unburnt, seed, fertiliser level 1
BSF2	burnt, seed, fertiliser level 2	USF2	unburnt, seed, fertiliser level 2

Fertiliser treatment

Superphosphate can enter native vegetation as windblown material (Muir 1979), and the fertiliser treatment imitates this event. Although most fertilisers currently in use are granular rather than powdered, the process is incomplete, so a proportion of fine material (from powder to small particles) would still be able to drift. The fertiliser amounts were based on the recommended levels for improved pasture in the surrounding agricultural region. More specifically, the amounts were based on the recommended pasture maintenance levels for stock production, not the levels recommended for pasture establishment or hay cutting which are generally much higher (N. Mendham, Agricultural Science, University of Tasmania, *pers. comm.* 1996; G. Richardson, Impact Fertilisers, *pers. comm.* 1996).

Previous experiments on fertiliser additions in heathlands found phosphorus and nitrogen to be most critical (Specht 1963). This experiment focuses on phosphorus because it is the most common pasture fertiliser, whereas nitrogen is generally provided by sowing legumes. Therefore the fertilisers used were superphosphate, with the addition of muriate of potash, sodium molybdenite and copper sulphate to simulate the local fertilising conditions. The timing of the applications was based on the usual practise for sandy soils: halving the amount of super and adding one dose in spring with the potash and one dose in autumn with the trace elements.

Fertiliser level 1 (BF1, BSF1, UF1, USF1) was one quarter the agricultural amount and level 2 (BF2, BSF2, UF2, USF2) was one half. These amounts were chosen to simulate fertiliser drift. The fertiliser was broadcast by hand over an area of 5 x 5 m, which included the 4 x 4 m quadrat plus half a metre further out all around. The spring application was delayed until after the burn and occurred on 19–20 December 1996, and the autumn application occurred over 30 May – 8 June 1997.

The amount of fertiliser was quite small (Table 3.3), and needed to be mixed with a filler to enable it to be spread evenly. The super and potash were ground in a mortar and pestle and sieved to 2 mm, the trace elements required no processing. The filler was washed and

steam-cleaned coarse river sand sieved to 2 mm and dried in a 100°C oven for 5 hours. The fertilisers and sand were weighed, bagged, shaken and broadcast by hand.

Table 3.3. Amounts of fertiliser applied per 5m x 5m quadrat

season of application	fertiliser	level 1 (g/25m^2)	level 2 (g/25m^2)
spring	super	31.25	62.50
	potash	31.25	62.50
autumn	super	31.25	62.50
	molybdenum	0.05	0.10
	copper	5.50	11.00
total amount	**fertiliser**	**level 1 (kg/ha)**	**level 2 (kg/ha)**
	super	25.00	50.00
	potash	12.50	25.00
	molybdenum	0.02	0.04
	copper	2.25	4.50

Fire treatment

A spring burn was the most desirable for the fire treatment for two reasons. Firstly, it would be optimal weather for fire control and, secondly, a spring burn would allow the longest monitoring period, given the time constraints of the project. The availability of the fire crew and the wait for safe weather led to the burn occurring in early summer.

Burn blocks were prepared by putting in firebreaks using a heavy-duty whipper snipper. Due to the configuration of tracks, the number of quadrats and a misunderstanding by the local fire brigade, three separate blocks were burnt (Figure 3.1). Block 2 was burnt on 14 November 1996 with an evening fire that did not burn strongly enough. The quadrats were unusable because the vegetation was not entirely burnt and was also crushed and disturbed by raking. These were replaced with more quadrats and surrounded by a new firebreak in block 3. Blocks 1 and 3 were burnt on 5 December 1996, commencing at 1.00 p.m. and finishing at 2.30 p.m. (Figure 3.3b). Thirty of the sixty quadrats were burnt. The fuel moisture was very constant at 13.1%. The burn was a good imitation of a natural burn with a fire front that progressed at 4.7 m/min, and a residence time of approximately half a minute.

Exotic grass seed treatment

The seeds used were two pasture grasses: *Dactylis glomerata* var. Porto (Cocksfoot Porto) and *Lolium perenne* var. Jackaroo (Ryegrass Jackaroo). These grass seeds were chosen because they are commonly used and do not have rhizomes, making them relatively easy to remove at the end of the experiment. As with the fertilisers, the amount was half the

recommended sowing amount for improved pasture in the surrounding agricultural area (Table 3.4).

Table 3.4. Amounts of exotic grass seed added per 5 m x 5 m quadrat

grass seed	quadrat (g/25m^2)	total (kg/ha)
Cocksfoot	2.5	1
Ryegrass	15	6

The two seed types were weighed and mixed together in sample bags and broadcast by hand. They were applied over a 5 m x 5 m area including the 4 m x 4 m quadrat, as for the fertiliser treatment. Germination damage can occur from copper and potash so time was allowed for some rain between the fertiliser treatment and the seed application. The seeds were applied three weeks after the fertiliser treatment, on 15 January 1997. Rainfall for this period was adequate for germination (Figure 3.2b).

Sampling design and measurements

Vegetation

Data for three species–area curves was collected by identifying all the species in a 50 cm square, then doubling the size and counting the number of new species, and continuing in this way until no new species were counted for the last two increases (Mueller-Dombois and Ellenberg 1974). The three curves all flattened out at approximately 16 m^2, so the quadrats used were 4 m x 4 m squares.

Each quadrat was allocated a random number and plotted on a grid over the study site. Quadrats falling in the wrong vegetation type were discarded and new quadrats plotted, until there were 60 quadrats in the heath area. In the field, quadrats were sited according to the grid location and marked using 20 cm long metal deck spikes in each corner.

All quadrats to be burnt were matched with the unburnt quadrats, using Bray-Curtis dissimilarity scores based on cover abundance values. These matched pairs were used during data analysis to help counteract the potential problem of site effects due the grouping of burnt quadrats in blocks. The fertiliser and weed seed treatments were randomly allocated to the unburnt quadrats and the same treatment was given to the burnt pair. The 14 inappropriately burnt quadrats were replaced with 14 new quadrats and rematched with their unburnt pairs before the experimental burn took place.

The 60 quadrats were visited once in every season (except winter) over a two-year period, resulting in seven scores (Table 3.5). Slope and aspect were recorded for all quadrats in the first visit. For the first and subsequent visits, identifiable vascular plant taxa were

recorded and given a cover class (1, <1%; 2, 1–5%; 3, 6–25%; 4, 26–50%; 5, 51–75%; 6, > 75%). The species that were dominant and subdominant for the overstorey and the understorey were noted. A species was considered dominant if had a cover of greater than 50%, and subdominant if it had a cover of at least 50% that of the dominant species. The perimeter of the quadrat was used as a 16 m line intercept to measure the percentage cover of live plants, dead plants and bare ground. Species nomenclature follows that of Buchanan (1999).

Table 3.5. Summary of timing of treatments and measurement periods

date	treatment	vegetation score	scats and soil
June 1996		species–area curves	
late Aug 1996		Spring 1	
Sep 1996			first soil samples
early Nov 1996		Spring 1 replacement quads	
early Dec 1996	BURN		
late Dec 1996	fertiliser spring		
mid Jan 1997	exotic grass seeds		
Feb/Mar 1997		Summer 1	
March 1997			first scat count and removal
May 1997	fertiliser autumn	Autumn 1	scats and diggings
Oct 1997		Spring 2	scats and diggings
Feb 1998		Summer 2	scats and diggings
May/June 1998		Autumn 2	scats and diggings
Sep 1998		Spring 3	scats and diggings last soil samples
Oct 1999		Spring 4	scats and diggings

Three quadrats were slashed accidentally before the Summer 2 score when a firebreak was put in as part of the fire management plan for the Nature Reserve. Species presence/absence could still be scored, but the line intercept became meaningless. It was too late in the experiment to replace these quadrats. One year after the end of the experiment (Spring 4), all quadrats were checked for *Hypochoeris radicata* seedlings, and 12 temporary, randomly located control quadrats (6 burnt and 6 unburnt) were included in this score.

Soils

Soil samples were taken at the beginning and end of the project. A 45-mm diameter metal soil corer was used to take samples at 0–5 cm and 5–10 cm depths. Five samples were taken per 4 x 4 m quadrat and bulked in the field. The first samples were taken over the period 26 November – 5 December 1996, and the second samples over 10 – 18 September 1999. Samples were always taken from under shrubs and during similar rainfall conditions. The bagged

soil samples were opened and left to air dry for two weeks, then ground and sieved to 500 µm. Phosphorus was analysed by Bray's extractable phosphate method using an acid ammonium fluoride extractant (method 2) and total nitrogen using Kjedahl digestion (Jackson 1958).

Scats

Native and exotic animal grazing was assessed by counting scats in 80 cm x 4 m quadrats along two parallel sides within the vegetation quadrat. In order to establish a baseline, in the first scat count (Table 3.5) the scats were removed but not counted. After that, the scats were identified as either rabbit or native animal and counted and removed during each vegetation score. Native animals included wallaby, pademelon, eastern-barred bandicoot, long-nosed potoroo, possum and native rat (Haseler 1994). Native diggings (by potoroos and bandicoots) were scored by counting each discrete conical hole as one digging, excluding shallow rabbit scrapes. Diggings had to be fresh since the last score, with few fallen leaves and no seedlings in the hole or surrounding soil.

Statistical analysis

One-way ANOVAs were used to test for differences between the treatments within each season. Data were assessed as normal or non-normal by testing the residuals using the Anderson-Darling statistic. Where transformation was needed, either natural log, square root, arcsine or the lambda value suggested by a Box-Cox transformation was used, depending on the type and shape of the data (Sokal and Rohlf 1995). The Box-Cox transformation estimates a lambda value that minimises the standard deviation of the transformed variable. The transformation with the most normal residual was then used, meaning that different variables and different scores therefore sometimes had different transformations. Variables that could not be transformed to normality were testing using the Kruskal-Wallis rank order test as a non-parametric alternative to ANOVA. Burnt and unburnt treatments were dealt with separately (except where indicated below).

Fertiliser and seed treatments were grouped in three different ways depending on the purpose of the analysis (see Table 3.2 for key to symbols):

1) differences between all five treatments: BC, BF1, BF2, BSF1, BSF2, for burnt treatment, UC, UF1, UF2, USF1, USF2, for unburnt treatment;
2) differences between three fertiliser levels: BC (no fertiliser), BF1/BSF1 (fertiliser level 1), BF2/BSF2 (fertiliser level 2) for burnt treatment, same for unburnt treatment;
3) differences between presence and absence of exotic grass seeds: BC/BF1/BF2 (no seed), BSF1/BSF2 (seed added) for burnt treatment, same for unburnt treatment.

Differences between fertiliser and seed treatments were also tested by subtracting burnt and unburnt pairs from each other and testing the resultant differences as above. Differences between treatments in change over time was investigated by subtracting the value of a variable in the second season from the same variable in the first season (Spring 2 – Spring 1, Spring 3 – Spring 2, Spring 3 – Spring 1, Summer 2 – Summer 1 and Autumn 2 – Autumn 1).

Two-way ANOVAs were used to test for interactive differences between fertiliser and seed (for burnt and unburnt separately) and between fertiliser and burning (addition of seed was ignored). For live cover and bare ground the data were not normal or able to be transformed to normality and the Friedman Test for a randomised block design was used. The Kruskal-Wallis Test for one-way design was used to check for significant differences between the burnt and unburnt treatments.

To look at the effect of fertilising and burning on changes in the cover of individual plants, cover 'transitions' were counted. A transition was a change in cover class of either increased, decreased or remained the same. There were six transitions between the seven time periods. There were 36 transitions for a plant species for both the burnt and unburnt controls (6 quadrats x 6 transitions), and 72 transitions each for the burnt fertiliser level 1 and 2, and unburnt fertiliser level 1 and 2 treatments (12 quadrats x 6 transitions). Quadrats that had no appearance of the species in any time were excluded, and species that appeared in less than half the quadrats were excluded. The cover transitions were tested for significant differences by using the Chi-squared test. The Yates correction for continuity was used where d.f. = 1. Cases where expected values where less than 5 (for d.f. = 1) or less than 4 (for d.f. = 2) were excluded (Zar 1996). The Chi-squared test with the Yates correction was also used to test for significance in the frequency of exotic herbs, exotic grasses and native grasses, and for the number of rabbit or native animal scats and native animal diggings.

Correlations between soil nitrogen levels and shrub cover were tested using Pearson's Product Moment Correlation Coefficient for parametric data and Spearman's Rank Correlation Coefficient for non-parametric data. The first set of soil samples was tested against shrub cover for the first score (Spring 1), and the last set of soil samples was tested against shrub cover for the last score (Spring 3). The analyses were done using the statistical package MINITAB (Release 12.2).

RESULTS

Native plant cover

There were no significant differences in dead cover, live cover or bare ground between all

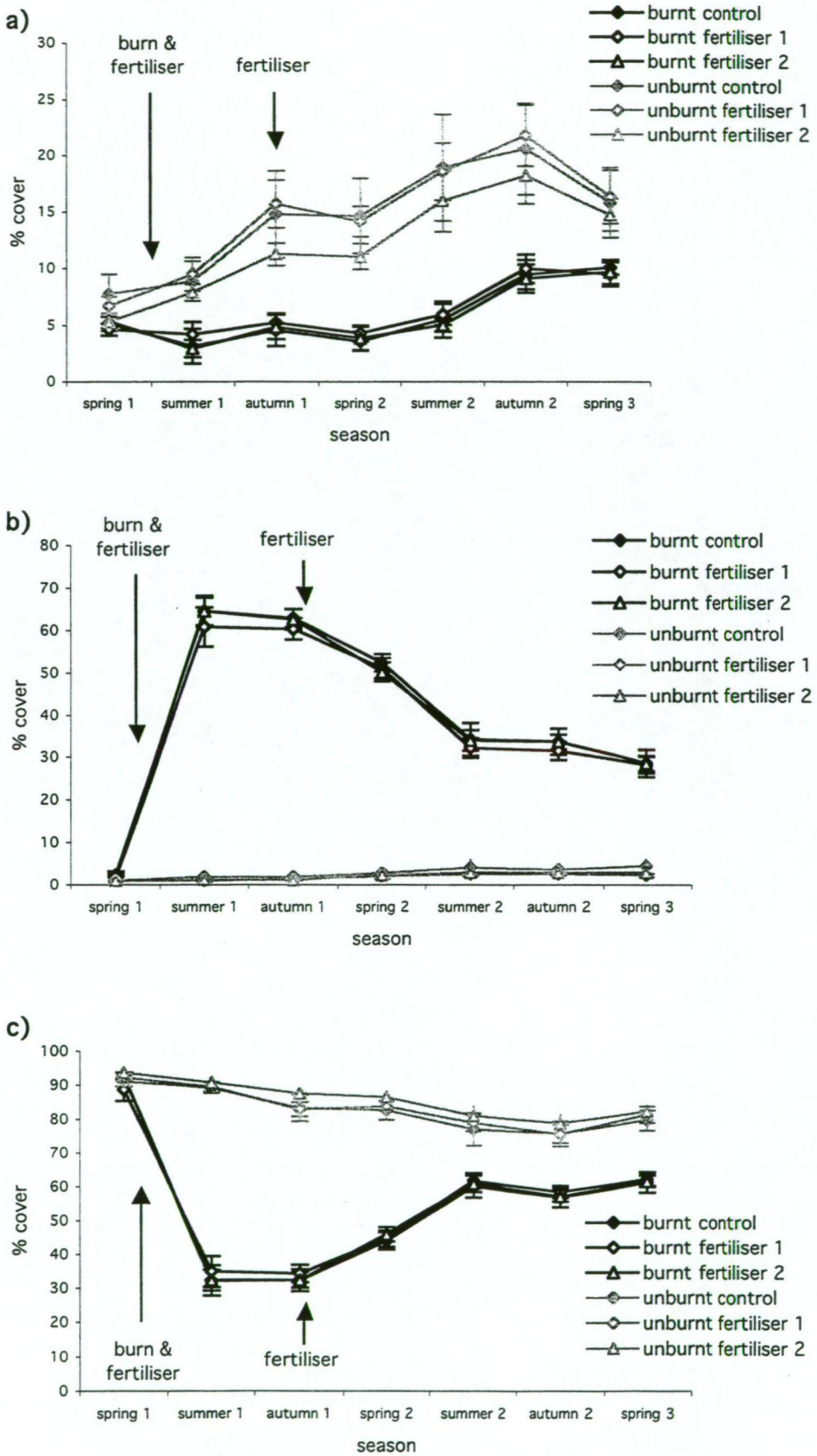


Figure 3.4. Average percentage cover of a) dead plants, b) live plants and c) bare ground over time. Standard error bars are shown. Arrows indicate when treatments were applied

five treatments, between the three fertiliser levels or between exotic grass seed or no seed (Figure 3.4a–c). This was the case for both burnt and unburnt treatments and for all seven seasons. There were also no significant differences in dead cover, live cover or bare ground between the fertiliser or seed treatments in change over time, or in burnt-unburnt paired quadrats. Dead cover and bare ground tended to increase over the study period and live cover tended to decrease in the unburnt quadrats. Dead cover and live cover were significantly higher in the unburnt treatment compared with the burnt treatment, for all seasons after the burn had taken place. Bare ground started at around 1% and was significantly higher in the burnt treatment both before the burn took place ($P < 0.05$) and for every season after the burn. By the end of the experiment bare ground had increased to an average of 2.4% in the unburnt quadrats and dropped from 65% to 30% in the burnt quadrats. There were no interactive effects between fertiliser and exotic grass seed or between fertiliser and fire for dead cover, live cover and bare ground.

Species richness

There were no significant differences in species richness between all five treatments, between the three fertiliser levels or between presence or absence of exotic grass seeds. This was the case for both burnt and unburnt treatments and for all seven seasons (Figure 3.5). There were also no significant differences when comparing burnt quadrats with their unburnt pairs or when looking at changes over time. Species richness was significantly lower in Summer 1 ($P < 0.001$) and Autumn 1 ($P < 0.01$) in the burnt treatment compared to the unburnt treatment, but from Spring 2 onwards there were no significant differences in richness between burnt and unburnt treatments (Figure 3.5). There were no interactive effects of fertiliser and fire for any season.

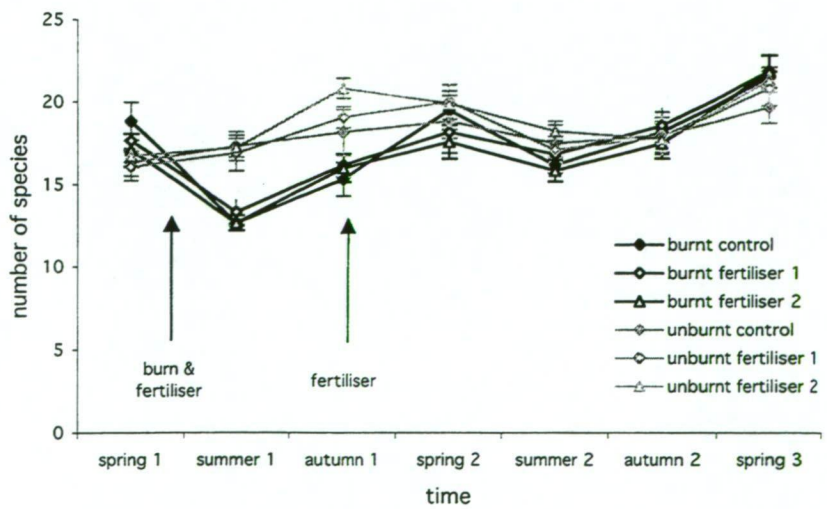


Figure 3.5. Average species richness over time by burning treatment and fertiliser level. Arrows indicate when treatments were applied. Standard error bars are shown

Exotic grasses and herbs

No exotic grasses were found in quadrats that were not sown with seed. Exotic grasses germinated in the first autumn after they were sown and then decreased rapidly until none were present by the end of the experiment. The difference in frequency of exotic grasses was significant in Autumn 1, with more quadrats having grasses if they were sown with seed (Figure 3.6b). The frequency of exotic grasses was different between the two levels of fertiliser (Figure 3.6a). This was only weakly significant ($P < 0.05$), with more fertiliser level 2 quadrats having exotic grasses than expected (10 out of 24) in Autumn 1. There was no difference in the frequency of exotic grasses between burnt and unburnt treatments (Figure 3.6c).

Only one exotic herb, *Hypochoeris radicata*, was found over the course of the study. Although this exotic herb was only found in fertilised quadrats during the experiment, it was not found in enough quadrats for this difference to be significant. Over the period of the experiment the frequency of *H. radicata* increased. However, when the site was revisited one year after the end of the experiment, the frequency had decreased and *H. radicata* was found equally in fertilised and control quadrats (Figure 3.7a). More burnt than unburnt quadrats had *H. radicata* seedlings but, again, the difference was not significant (Figure 3.7b).

Native lifeforms

Shrubs

There were no significant differences in shrub species richness between fertiliser levels for in any season, for either burnt or unburnt treatments. Shrub species richness was significantly higher in the unburnt treatment than the burnt treatment for all seasons after the burn (Figure 3.8a). Shrub abundance followed the same pattern, with significantly higher abundances in the unburnt treatment for all seasons after the burn (Figure 3.8b).

Four shrub species had significant differences in cover transitions between fertiliser levels or the fire treatment. *Aotus ericoides* had more stable cover (cover classes remained the same over the study period) in the unburnt control with less increases and decreases than expected, compared with the unburnt fertilised quadrats (fertiliser level 2) (Table 3.6). The same was true when the unburnt control was compared with the burnt control.

Baeckea ramosissima behaved in a similar way to *Aotus ericoides*. Although there were no significant differences between the unburnt control and fertilised quadrats, the differences were significant when the fertiliser levels were compared separately. Cover values decreased more often than expected in unburnt fertiliser level 1 and less often in the

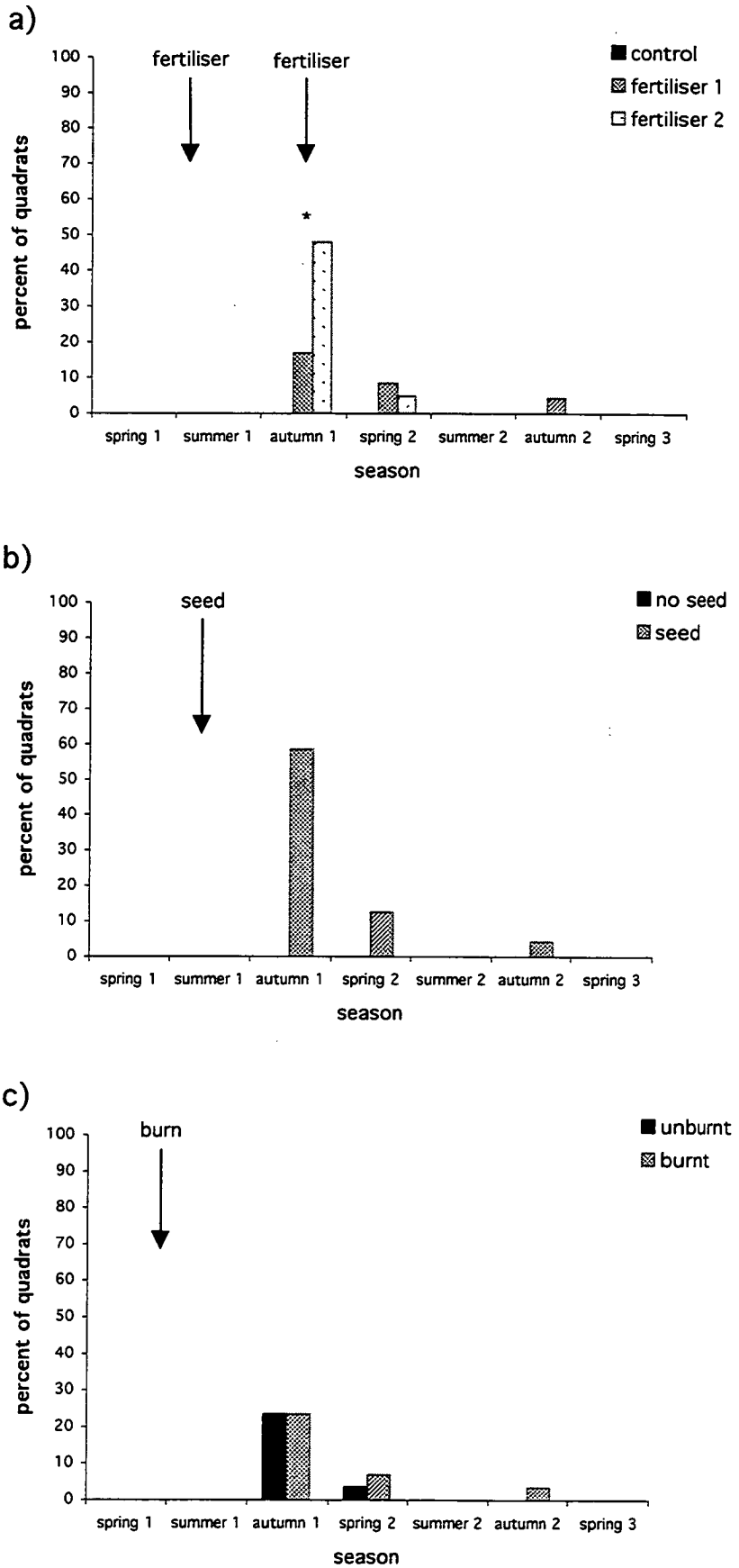


Figure 3.6. Frequency of exotic grasses by a) fertiliser level, b) seed treatment and c) burn treatment. Arrows indicate when treatments were applied. *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

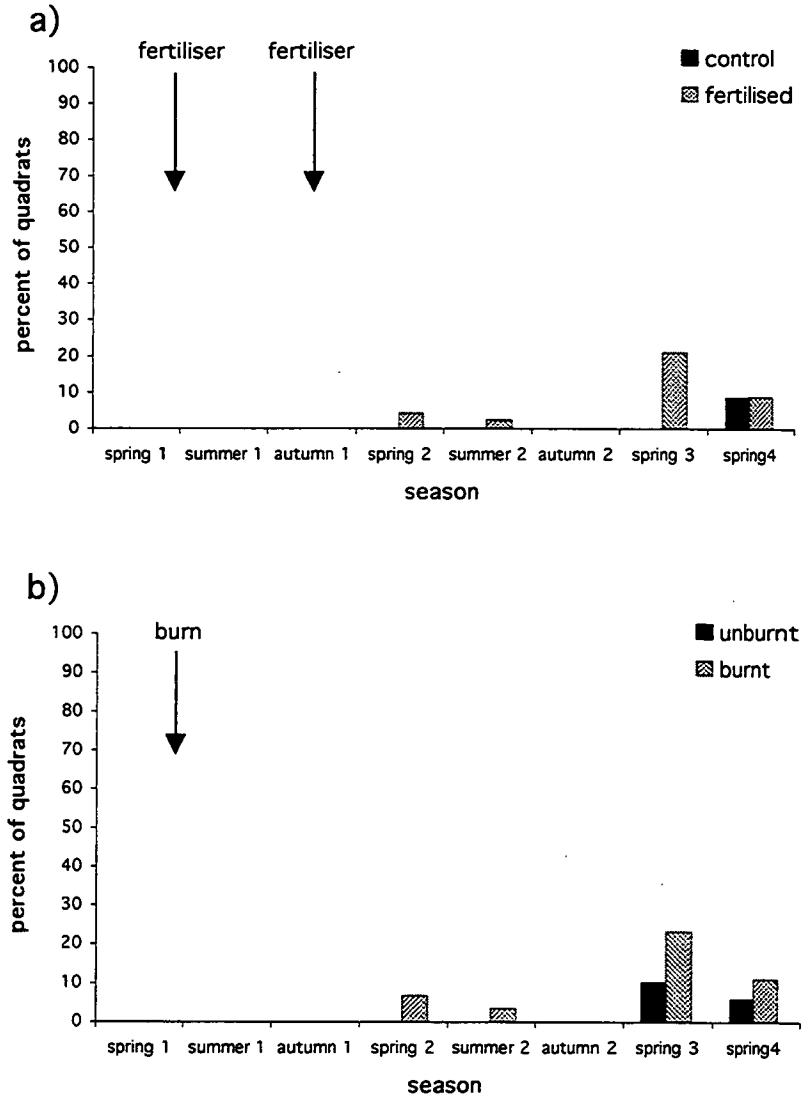


Figure 3.7. Frequency of exotic herbs by a) fertiliser treatment and b) burn treatment. Arrows indicate when treatments were applied

Table 3.6. Summary of chi squared significance levels for shrub cover transitions for BC/BF, burnt control compared with burnt fertilised; BC/BF1/BF2, burnt control compared with burnt fertiliser level 1 compared with burnt fertiliser level 2; UC/UF, unburnt control compared with unburnt fertilised; UC/UF1/UF2, unburnt control compared with unburnt fertiliser level 1 compared with unburnt fertiliser level 2; BC/UC, burnt control compared with unburnt control; BF/UF, burnt fertilised compared with unburnt fertilised

	BC/BF	BC/BF1/BF 2	UC/UF	UC/UF1/UF 2	BC/UC	BF/UF
Amperea xiphoclada	*	*	ns	ns	ns	ns
Aotus ericoides	ns	ns	**	**	*	ns
Baeckea ramosissima	ns	ns	ns	*	ns	*
Bossiaea cinerea	ns	**	ns	ns	ns	ns
Epacris impressa	—	—	ns	ns	—	—
Hibbertia acicularis	ns	ns	ns	ns	ns	ns
Leptospermum scoparium	ns	ns	ns	ns	ns	ns
Leucopogon collinus	—	—	ns	ns	—	—
Leucopogon ericoides	—	—	ns	ns	—	—
Pimelea linefolia	—	—	ns	ns	—	—
Styphelia adscendens	—	—	ns	ns	—	—

—, too few plants to test; ns, not significant; *, P< 0.05; **, P< 0.01; ***, P< 0.001

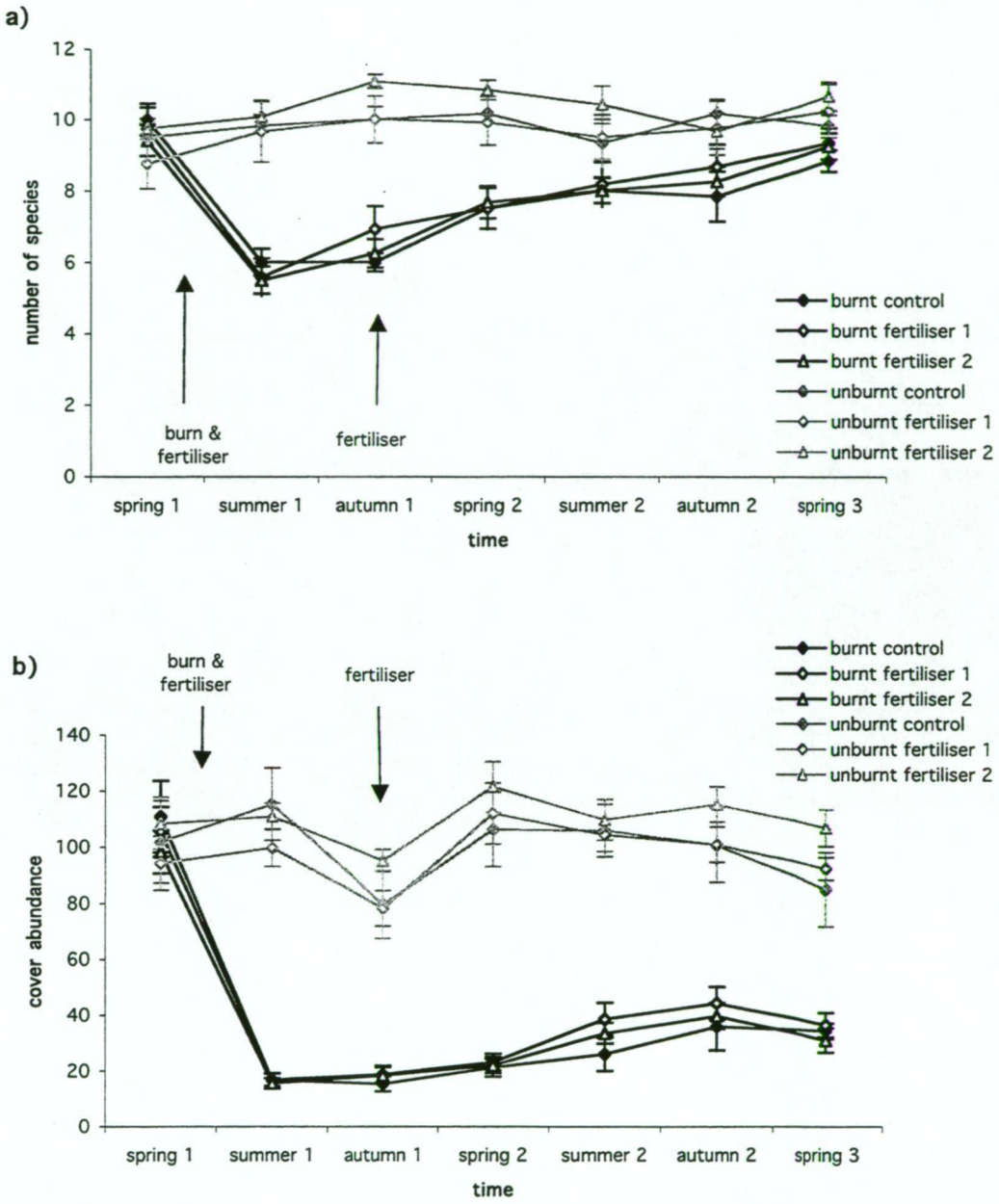


Figure 3.8. Shrub a) species richness and b) abundance by fertiliser level. Standard error bars are shown. Arrows indicate when treatments were applied

unburnt control. When burnt and unburnt fertilised treatments were compared, unburnt had more decreases than expected, and burnt had fewer decreases than expected. This is opposite to the *A. ericoides* result in which the unburnt treatment was more stable. However for *A. ericoides* the significant difference was between the control quadrats, whereas for *B. ramosissima* it was for fertilised quadrats.

Bossiaea cinerea also showed significant differences in cover values over time. In this case the differences were in the burnt treatment — cover values had fewer increases than

expected in fertiliser level 1 and more increases than expected in burnt control. This result runs counter to that of *A. ericoides* and *B. ramosissima,* where there was more stability in unburnt control quadrats and less stability for unburnt fertilised quadrats. *Amperea xiphoclada* cover had less increases than expected in the burnt control treatment, but had more increases than expected in the burnt fertilised treatment. This was related to differences in fertiliser level 2.

Grasses

The frequency of native grasses was higher in burnt quadrats than unburnt quadrats. This result was highly significant ($P< 0.001$) in Spring 2 and weakly significant ($P< 0.05$) in Autumn 2. Other than that there were no significant differences in the number of quadrats between control and fertilised, different levels of fertiliser, or presence or absence of exotic grass seeds (Figure 3.9).

Herbs

One herb, *Gonocarpus tetragynus* (Common Raspwort), had significant differences in cover values between burnt and unburnt. Cover was less stable in the unburnt control with more increases and decreases than expected compared with the burnt control. The same was true for fertilised quadrats; cover was less stable in unburnt fertilised and more stable in burnt fertilised (Table 3.7).

Table 3.7. Summary of chi squared significance levels for herb cover transitions. For description of codes see Table 3.6

	BC/BF	BC/BF1/BF2	UC/UF	UC/UF1/UF2	BC/UC	BF/UF
Cassytha glabella	ns	ns	ns	ns	ns	ns
Drosera species	ns	ns	ns	ns	ns	ns
Gonocarpus tetragynus	ns	ns	ns	ns	***	***
Helichrysum scorpioides	ns	ns	—	—	—	—
Stylidium graminifolium	ns	ns	ns	ns	ns	ns

—, too few plants to test; ns, not significant; *, $P< 0.05$; **, $P< 0.01$; ***, $P< 0.001$

Graminoids

There were no significant differences in graminoid species richness between fertiliser levels for any season, when burnt or unburnt treatments were looked at separately. When burnt and unburnt treatments were compared, there were also no significant differences. However, graminoid abundance was significantly lower in the burnt quadrats for four seasons after the burn, after which it was not significantly different from the unburnt

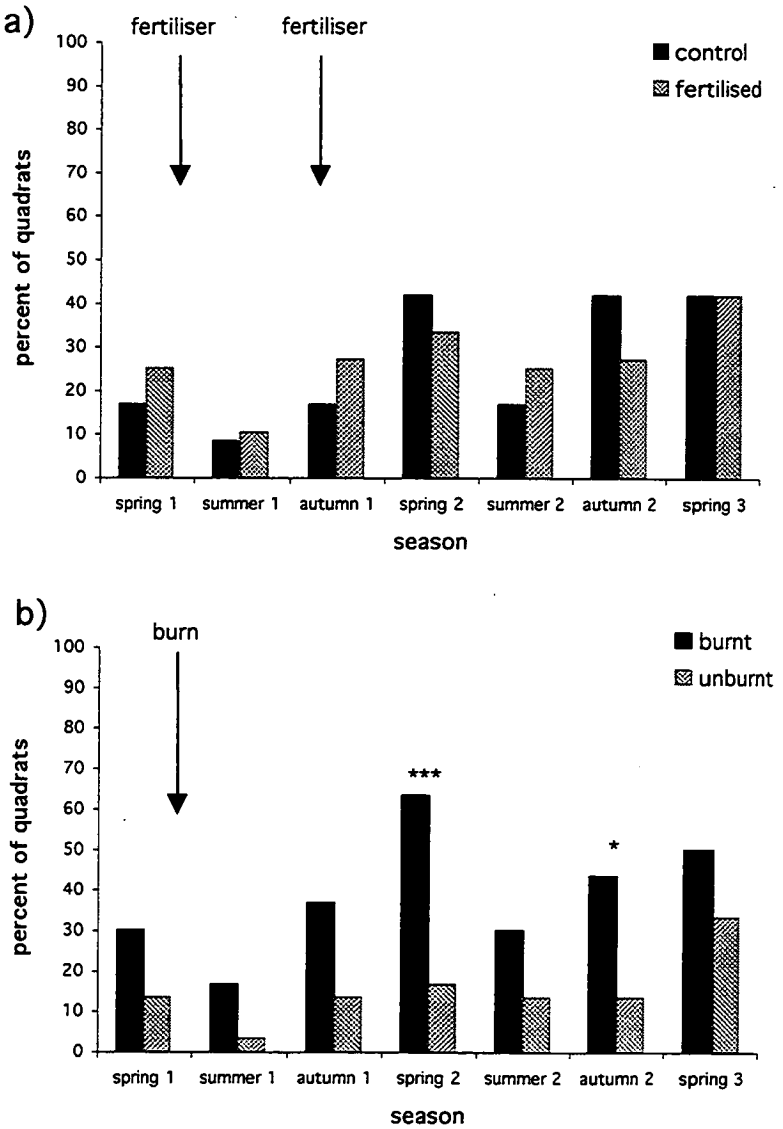


Figure 3.9. Frequency of native grasses by a) fertiliser treatment and b) burn treatment. Arrows indicate when treatments were applied. *, P< 0.05; **, P< 0.01; ***, P< 0.001

quadrats (Figure 3.10). One graminoid species had significant differences in cover transitions (Table 3.8). Cover values for *Hypolaena fastigiata* had fewer decreases than expected in unburnt fertiliser level 1 and many more decreases than expected in unburnt level 2. When unburnt fertiliser level 1 and level 2 were combined and compared with the unburnt control, there were no significant differences in cover transitions. *Lepidosperma*

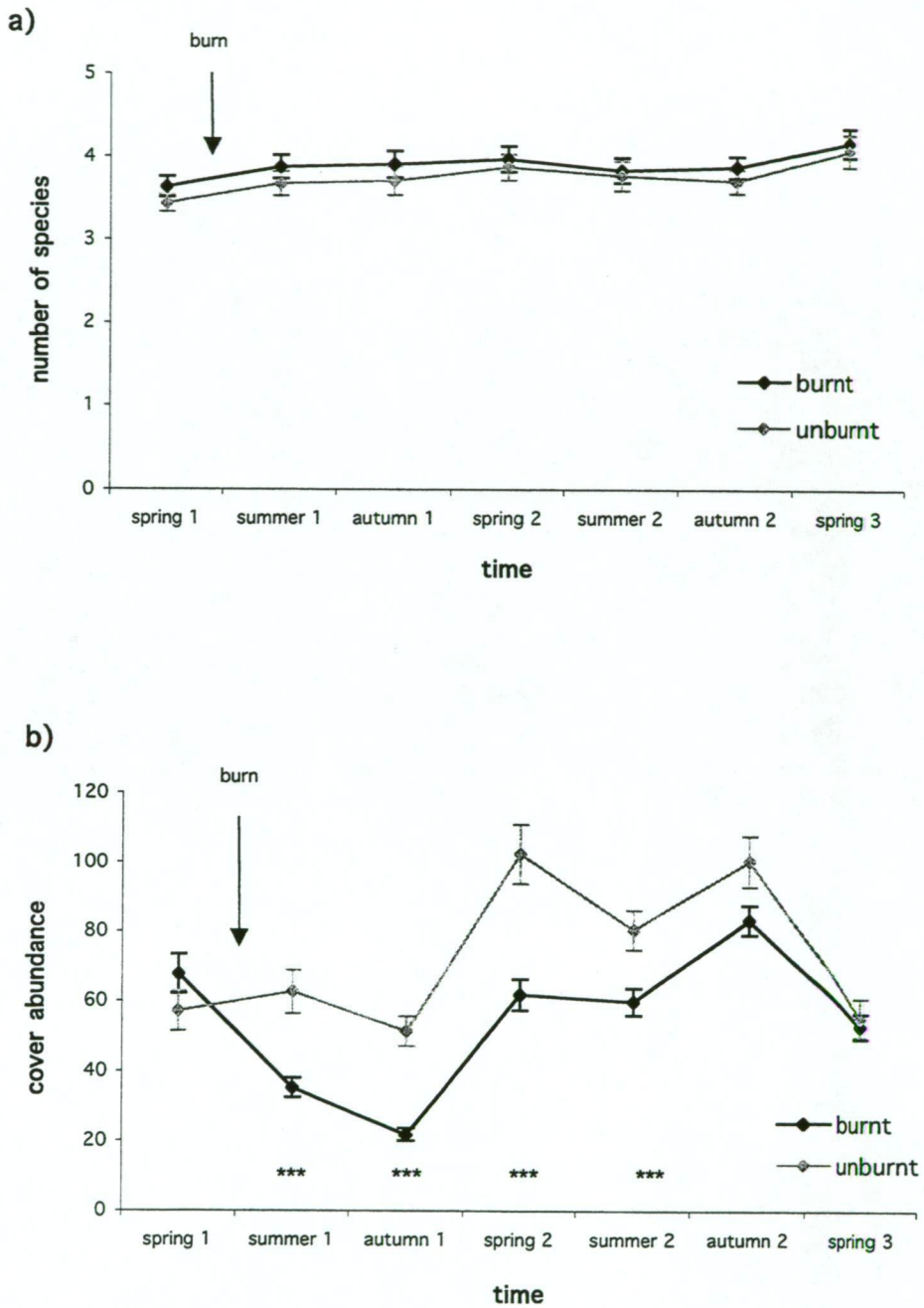


Figure 3.10. Graminoid a) species richness and b) abundance by burn treatment. Standard error bars are shown. Arrows indicate when treatments were applied. *, P< 0.05; **, P< 0.01; ***, P< 0.001

filiforme (Thread Rapier-sedge) showed a more stable cover over the study period in the burnt control compared with the burnt fertilised treatment.

Table 3.8. Summary of chi squared significance levels for graminoid cover transitions. For description of codes see Table 3.6

	BC/BF	BC/BF1/BF2	UC/UF	UC/UF1/UF2	BC/UC	BF/UF
Hypolaena fastigiata	ns	ns	ns	*	ns	ns
Lepidosperma filiforme	**	**	ns	ns	ns	ns
Schoenus lepidosperma	ns	ns	ns	ns	ns	ns

ns, not significant; *, P< 0.05; **, P< 0.01; ***, P< 0.001

Rabbit and native animal scats

The number of rabbit scats was higher in the burnt treatment than the unburnt treatment, for every season after the fire. There were no significant differences in the number of rabbit scats between fertilised and not fertilised or between presence or absence of exotic grass seeds (Figure 3.11a).

The number of native animal scats was also significantly higher in burnt quadrats than unburnt quadrats, for Autumn 1, Spring 2 and Summer 2. There were no significant differences in native animal scats between fertilised and not fertilised or between presence or absence of exotic grass seeds (Figure 3.11b).

Native animal diggings showed the opposite pattern. There were significantly more diggings in unburnt quadrats in Spring 2 and Summer 2. There were no significant differences between diggings in burnt and unburnt quadrats after Summer 2. Again, there were no significant differences in native animal diggings between fertilised and not fertilised or between presence or absence of exotic grass seeds (Figure 3.11c).

Soil nutrients

At the beginning of the experiment there were no differences in phosphorus levels in the top or bottom soil layers between the treatments (before they were applied). However, there were also no significant differences between fertilised and unfertilised treatments at the end of the experiment (Table 3.9). Nevertheless, there were some significant differences in nutrient levels between the start and end of the experiment.

When all treatments were taken together, P top (phosphorus in top 0–5 cm of soil) and P bottom (phosphorus in 5–10 cm depth of soil) were significantly higher in Time 2 (at the end of the experiment) compared with Time 1 (at the beginning of the experiment). When burnt and unburnt treatments were tested separately, P top and P bottom were always higher in Time 2. When fertiliser levels were also separated, P top was higher in Time 2 for BC, BF1, BF2, UF1 and UF2 and P bottom was higher in Time 2 for BF1, BF2, UC and

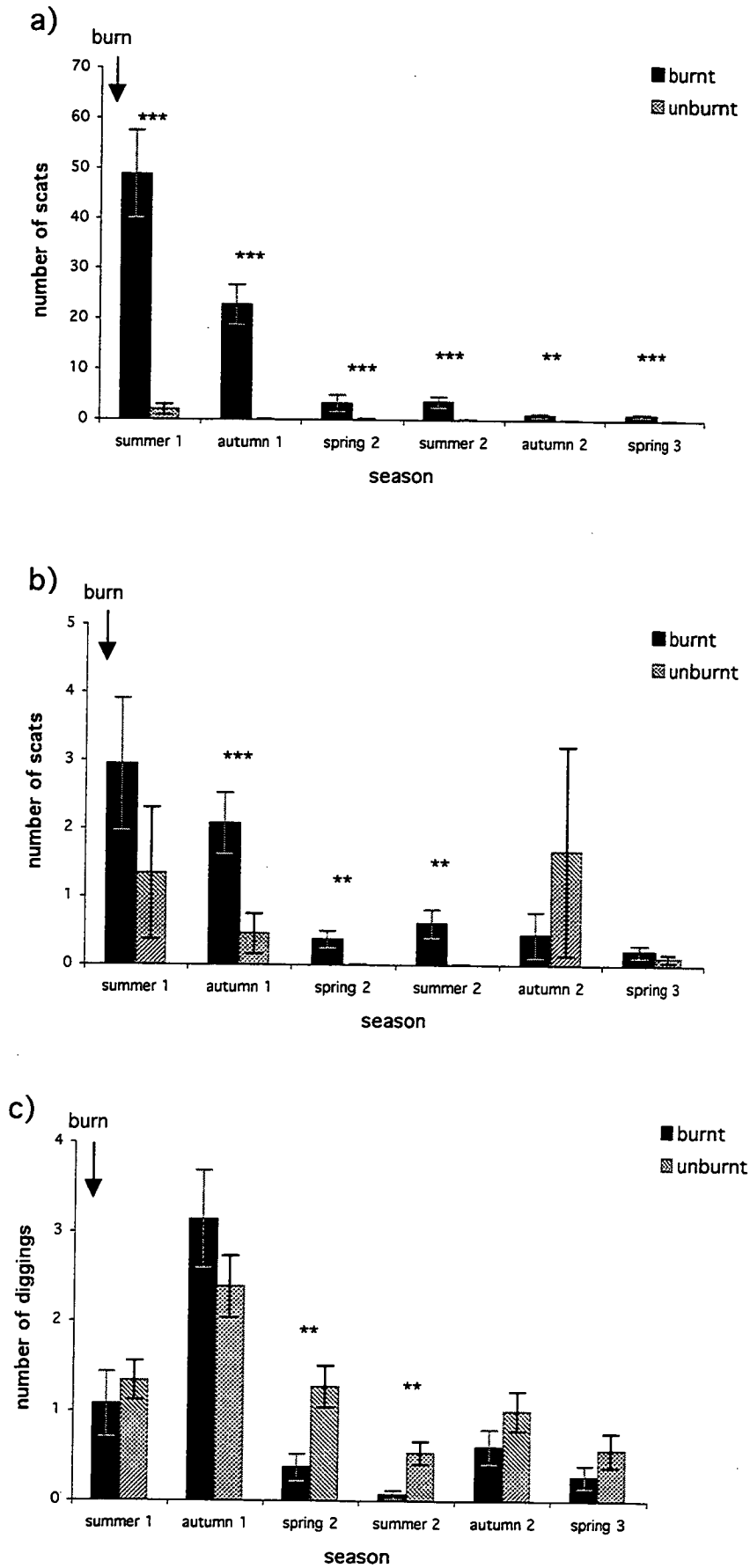


Figure 3.11. Number of a) rabbit scats, b) native animal scats and c) native animal diggings. Standard error bars are shown. Arrows indicate when treatments were applied. *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

UF2. That is, phosphorus levels were higher in Time 2 even in the control quadrats, with P top being higher in BC and P bottom being higher in UC. Comparisons were also made between top and bottom phosphorus samples. When all times and treatments were combined there were no significant differences between top and bottom samples. However, P top was higher than P bottom for unburnt quadrats in Time 2.

Table 3.9. Summary of results for soil phosphorus and nitrogen, for control compared with fertilisers, Time 1 compared with Time 2, and the top soil layer compared with the bottom layer

phosphorus	control	fertiliser	P	nitrogen	control	fertiliser	P
top time 1	1.17 (0.11)	1.07 (0.04)	ns	top time 1	0.170 (0.018)	0.133 (0.005)	*
bottom time 1	1.11 (0.10)	1.23 (0.20)	ns	bottom time 1	0.145 (0.013)	0.122 (0.006)	ns
burnt top 1	1.34 (0.15)	1.02 (0.04)	ns	burnt top 1	0.147 (0.021)	0.135 (0.006)	ns
burnt bottom 1	1.23 (0.09)	1.35 (0.41)	ns	burnt bottom 1	0.125 (0.016)	0.117 (0.008)	ns
unburnt top 1	0.99 (0.12)	1.13 (0.06)	ns	unburnt top 1	0.193 (0.028)	0.130 (0.009)	*
unburnt bot 1	1.01 (0.16)	1.11 (0.09)	ns	unburnt bot 1	0.165 (0.017)	0.128 (0.007)	*
top time 2	2.36 (0.31)	2.60 (0.12)	ns	top time 2	0.147 (0.015)	0.119 (0.006)	ns
bottom time 2	1.91 (0.31)	1.90 (0.15)	ns	bottom time 2	0.143 (0.011)	0.111 (0.006)	*
burnt top 2	2.60 (0.68)	2.71 (0.16)	ns	burnt top 2	0.141 (0.018)	0.118 (0.009)	ns
burnt bottom 2	2.60 (0.40)	1.79 (0.17)	ns	burnt bottom 2	0.126 (0.011)	0.115 (0.010)	ns
unburnt top 2	2.17 (0.17)	2.50 (0.18)	ns	unburnt top 2	0.154 (0.028)	0.121 (0.008)	ns
unburnt bot 2	1.33 (0.31)	2.02 (0.25)	ns	unburnt bot 2	0.167 (0.019)	0.107 (0.007)	*
phosphorus	**time 1**	**time 2**	**P**	**nitrogen**	**time 1**	**time 2**	**P**
all	1.15 (0.08)	2.24 (0.09)	***	all	0.134 (0.004)	0.121 (0.004)	**
top	1.09 (0.04)	2.56 (0.11)	***	top	0.140 (0.006)	0.125 (0.006)	*
bottom	1.20 (0.16)	1.90 (0.13)	***	bottom	0.127 (0.005)	0.117 (0.006)	ns
burnt top	1.01 (0.04)	2.60 (0.14)	***	burnt top	0.137 (0.006)	0.123 (0.008)	ns
burnt bottom	1.28 (0.32)	1.70 (0.15)	***	burnt bottom	0.118 (0.007)	0.117 (0.008)	ns
unburnt top	1.18 (0.06)	2.52 (0.18)	***	unburnt top	0.143 (0.010)	0.127 (0.008)	ns
unburnt bot	1.13 (0.08)	2.13 (0.22)	***	unburnt bot	0.136 (0.007)	0.117 (0.008)	ns
phosphorus	**top**	**bottom**	**P**	**nitrogen**	**top**	**bottom**	**P**
all time 1	1.09 (0.04)	1.20 (0.16)	ns	all time 1	0.140 (0.006)	0.127 (0.005)	***
all time 2	2.56 (0.11)	1.90 (0.13)	ns	all time 2	0.125 (0.006)	0.117 (0.006)	ns
burnt time 1	1.01 (0.04)	1.28 (0.32)	ns	burnt time 1	0.137 (0.006)	0.118 (0.007)	ns
burnt time 2	2.60 (0.14)	1.70 (0.15)	ns	burnt time 2	0.123 (0.008)	0.117 (0.008)	ns
unburnt time 1	1.18 (0.06)	1.13 (0.08)	ns	unburnt time 1	0.143 (0.010)	0.136 (0.007)	*
unburnt time 2	2.52 (0.18)	2.13 (0.22)	***	unburnt time 2	0.127 (0.008)	0.117 (0.008)	ns

ns, not significant; *, P< 0.05; **, P< 0.01; ***, P< 0.001

At the beginning of the experiment there were some differences in nitrogen levels between the different treatments, before the treatments were applied. Soil in the top 5 cm in the control quadrats had higher nitrogen levels than in the to-be-fertilised quadrats, in the unburnt treatment. Nitrogen in the 5–10 cm layer was also higher in unburnt control compared with unburnt fertiliser quadrats. There were no significant differences between treatments at the beginning or end of the study period in the burnt treatment. At the end of

the experiment, the bottom layer in the control quadrats still had higher nitrogen levels but the top layer was not significantly different. Generally, over the period of the experiment, the nitrogen levels declined, and this decline occurred in the top soil layer.

Nitrogen levels were significantly higher in the top compared with the bottom soil layers at the beginning of the experiment, when all treatments were combined. When burnt and unburnt treatments were separated, the top layer was significantly higher than the bottom layer in the unburnt treatment, Time 1. There were no differences between top and bottom layers in the burnt experiment.

When nitrogen was tested for correlation with shrub abundance, there was a significant correlation between N bottom and shrub abundance at the beginning of the experiment. There were no significant correlations at the end of the experiment.

DISCUSSION

Effects of fertilisers

Low levels of fertiliser application did not have an effect on the heathland vegetation at the Peter Murrell Nature Reserve over the study period. Dead cover, live cover, bare ground, total species richness, and shrub and graminoid species richness were not significantly different between control and fertilised quadrats, or between different levels of fertilisers. Neither were there differences in shrub, graminoid and herb species richness and abundance, nor in the frequency of native grasses. This lack of response of the vegetation explains the fact that there were no significant differences in rabbit or native animal grazing and native animal diggings between fertilised and control quadrats.

This experiment may help in identifying at what level nutrient inputs to heathland influence the vegetation. At Dark Island, 240–1940 kg/ha of superphosphate produced very obvious results. Mature heath plants increased growth and many species decreased density with increasing levels of super — annual growth increasing by up to four times in the second and third years after the addition of fertilisers (Specht 1963). Twenty-two years later, the positive growth response of the shrubs resulted in a more rapid life cycle and earlier death of the individual heath plants (Heddle and Specht 1975). In a heathland in Frankston, Victoria, 1000 kg/ha superphosphate and 1000 kg/ha blood and bone caused increased leaf area index and growth of heath species (Jones 1968). On North Stradbroke Island, Queensland, increased growth (measured as overlapping projective foliage cover) was found one year after application of a range of fertilisers including approximately 950 kg/ha of superphosphate (Specht *et al.* 1977). In the Netherlands, approximately 650 kg/ha of superphosphate was applied over three years, resulting in a decrease in cover of one shrub species (Aerts and Berendse 1988). In Beerwah, Queensland,

lower levels of approximately 400 kg/ha of super caused the density of most species to decrease, although the dry weight of some species increased and for other species decreased (Connor and Wilson 1967). Even at rates as low as approximately 150 kg/ha super, heath plants can show greatly increased growth (Clark 1975).

The levels of fertilisers in this experiment (25 and 50 kg/ha of superphosphate) were most likely to be too low to cause a measurable response, although two other experiments using comparably low levels of fertilisers still found that they caused changes to heathland vegetation. In a South African fynbos heath, Witkowski *et al.* (1990b) added phosphorus and nitrogen in amounts based on what would be returned to the soil and surface ash after a fire, equivalent to approximately 50 kg/ha of superphosphate (5 kg/ha P) and 50 kg/ha nitrogen. Two shrubs, *Leucospermum parile* (Malmsbury Pincushion) and *Phylica cephalantha* (Starface), showed increases in shoot growth and dry mass to nitrogen additions, although phosphorus additions either had no response or resulted in reduced shoot extension and dry mass in some cases. In a chaparral community in California, the amount of fertilisers added were approximately 40 kg/ha superphosphate and 80 kg/ha urea. The biomass of two shrubs, *Adenostoma fasciculatum* (Chamise) and *Ceanothus greggii* (Californian Lilac), increased most when nitrogen and phosphorus were added together, although *A. fasciculatum* responded most to nitrogen and *C. greggii* responded most to phosphorus (McMaster *et al.* 1982). These results indicate the importance of the relationship between nitrogen and phosphorus.

Imbalances between nitrogen and phosphorus explain the 'phosphorus toxicity' effect on the seedlings of many heath species at Dark Island and North Stradbroke Island (Specht 1963; Specht *et al.* 1977). In the cases where phosphorus increased growth, the addition of nitrogen with the phosphorus increased growth still further in some species. The short-term increased growth of nitrogen-fixing species also indicates that the N/P ratio is influencing the heathland species' responses (Heddle and Specht 1975). In a Colorado sagebrush community (found on slightly more fertile soils than true heathland) treated with fertilisers, there was no significant difference in above-ground biomass after fertilisation with phosphorus, although there were differences from nitrogen fertilisation (McLendon and Redente 1991). Berendse (1990) also found that changes in nitrogen supply have a significant effect on the competitive relationships between dominant heathland plant species. In the fynbos described above, Witkowski *et al.* (1990b) suggest their results could be explained by vegetative growth in some shrub species being limited by nitrogen, and give two theories as to why that might be so. Firstly, frequent burning might have depleted the nitrogen pool and, secondly, the lack of nitrogen input from rainfall in drier fynbos ecosystems may have caused imbalances in the N/P ratios. Australian heathlands generally have a greater proportion of nitrogen-fixing species than other heathland communities (Specht *et al.* 1958; Groves 1981b), and this is true for the heathland in the

present study. Therefore, lack of nitrogen is less likely to limit growth in the heathland in the Peter Murrell Nature Reserve. However, an imbalance between phosphorus and nitrogen could still be affecting growth patterns, particularly when the frequent burning is taken into account.

Although some plants respond very quickly to fertiliser additions (in the first two years for two chaparral shrubs (McMaster *et al.* 1982)), the apparent non-response of vegetation in this experiment may have been because there was not enough time for changes to become apparent. Changes may take three years before an increase in annual biomass is noticeable (Jones 1968; Hobbs and Atkins 1991), or much longer. Specht found that, due to extreme biological variation, changes in projective foliage cover that were obvious directly after fertilising were not statistically significant until nine years later (Specht 1963). Projective foliage cover of some species, e.g. *Banksia marginata* and *Xanthorrhoea australis* (Austral Grasstree), had no significant change even after nine years, although after 22 years *X. australis* decreased in density but not percentage cover and *Banksia marginata* increased in percentage cover but not density (Heddle and Specht 1975). Bare ground also showed no difference until 22 years later when there was a significant decrease in fertilised plots. Witkowski *et al.* (1990b) likewise suggest lack of time as a reason for why the two heathland plants they studied had such a small response to the addition of phosphorus.

Furthermore, some of the changes measured by the above authors were on a very fine scale, with detailed measurements being made of plant biomass, shoot growth and cover, etc. The broader measures of percentage cover and cover classes in the present study may not have picked up more subtle changes in plant growth or biomass. The constant death of bracken fronds would have confused the measurement of dead cover. It is also possible that plants were responding in ways that were not measured at all. For example, Witkowski *et al.* (1990b) hypothesise that one of the fynbos shrubs they studied may have been allocating fertiliser nutrient uptake to below-ground growth or storage.

Considerable differences in reaction to fertilisers, particularly phosphorus, are found from different species, as seen with the *Xanthorrhoea* and *Banksia* species above. In the present study, such differences are seen in the shrub *Aotus ericoides*, which had more fluctuating cover when fertilised, and the shrub *Baeckea ramosissima* which decreased more in cover when fertilised, indicating fluctuating growth and reduced growth, respectively. The shrub *Bossiaea cinerea* appeared to have slightly decreased growth when fertilised, as evidenced by less increases in cover than expected in burnt fertilised quadrats. The smaller shrub *Amperea xiphoclada* showed more cover stability when burnt compared with being burnt and fertilised and the graminoid *Hypolaena fastigiata* had more cover stability for the lower level of fertiliser addition when unburnt compared with no fertiliser or the higher fertiliser level.

Although the usual immediate response to the addition of fertilisers is an increase in shoot growth and the production of new leaves, this occurs most often in shallow-rooted herbaceous species (Pate and Dell 1982). High levels of nutrients have been found to increase native grass growth and species richness (Specht 1963; Hobbs and Atkins 1988), although in the present study no such response of native grasses was seen. Some large-seeded species in the families Proteaceae and Myrtaceae do not have an immediate response, for example *Leptospermum attenuatum* (Paperbark Teatree) and *Banksia aemula* (Wallum Banksia) (Specht *et al.* 1977). Other species not in either of these families are also unresponsive, for example *Xanthorrhoea australis* (Specht 1963), *Xanthorrhoea johnsonii* (Johnson's Grasstree) and *Allocasuarina littoralis* (Black She-oak) (Specht *et al.* 1977), at least in the short term. Some species experience 'phosphorus toxicity' due to an imbalance between phosphorus and nitrogen (Grundon 1972; Heddle and Specht 1975; Specht 1975; Groves and Keraitis 1976; Ozanne and Specht 1981). Obligate seeders and resprouters also tend to respond differently to fertilisers, particularly after fire — many plants that regenerate from rootstock have no change in density due to fertilisers (Specht 1963).

Such individualised responses to fertilisers would inevitably lead to a change in the balance of species in the heathland community (Pate and Dell 1982), leading in some cases to a change from the heathland to an herbaceous sward (Specht 1963; Heddle and Specht 1975). In this situation, the species that show least response to fertiliser additions become dominant in the community; other species showing large growth increases due to the addition of phosphorus are unable to dominate the community when phosphorus is lacking (Specht 1963). A return to the site of the present study in 10 or 20 years might show such changes and, if not, would confirm the hypothesis that small additions of phosphorus do not affect the vegetation community.

Effects of fire

Burning had a marked effect on the heathland, causing dead and live plant cover to be significantly reduced and bare ground to be significantly increased. Although in the present study live cover returned to approximately 70% of pre-burn levels after one year, and in another study to 88% in two years (Russell and Parsons 1978), generally heathland vegetation cover can take many years to return to pre-burn levels. Cover has been found to take more than four years (Posamentier *et al.* 1981), 5–6 years (Specht *et al.* 1958) and up to 10 years (Russell and Parsons 1978; Keeley *et al.* 1981) to return to the levels prior to a burn.

The burn in the present study was a reasonable simulation of a wildfire in late spring – early summer. However, it is likely to have been slightly cooler than a high intensity summer wildfire and therefore may have had some effect on the seed regeneration. Cooler

burns can reduce seed regeneration due to plants burning when in flower and with immature fruit, less seed being released or germinating due to lower intensity fires, increased mortality of seed due to higher temperatures in the season after fire (summer) and increased seed predation (Posamentier *et al.* 1981; Cheal 1996). Frequency of fires also affects the floristic composition of heathlands (Gill 1981; Gill and Groves 1981). High frequency fires (i.e. three fires in two years) were found to have a significant effect on floristic composition — frequency of some species being increased and others decreased by frequent fires (Bradstock *et al.* 1997). Frequent cool burns in the Peter Murrell Nature Reserve are probably responsible for the dominance of bracken (Duncan and Duncan 1995).

Heathlands generally experience an increase in species richness after fire. However there was no such increase over the period of the present study. The number of all species was reduced for two seasons after the burn, but thereafter was not significantly different. Species richness has been found to peak at 1 year (Keeley *et al.* 1981), overtake pre-fire levels after 1 year and peak at 4 years (Posamentier *et al.* 1981) or, more commonly, to be directly related to time since fire, declining over time (Specht *et al.* 1958; Russell and Parsons 1978). One reason species richness did not increase in the present study could be the lack of annuals found only after fire, a situation which also occurred at Wilson's Promontory (Russell and Parsons 1978), although not at Dark Island (Specht *et al.* 1958). Although the frequency of native grasses did increase significantly in some seasons after the fire, they were all species that were present before burning took place.

Different plant lifeform groups and species responded to being burnt in different ways, although there is not enough detail in the present study to assess these responses in terms of the heathland regeneration strategies described by various authors (Purdie 1977a; Purdie 1977b; Keeley *et al.* 1981; Mallik and Gimingham 1983; Keith and Bradstock 1994). Woody shrubs as a group responded differently compared with understorey and more herbaceous species, and different species within the lifeform group responded differently. Both shrub species richness and abundance remained significantly lower for all seasons after the burn, whereas for graminoids, even though total cover abundance was lower for four seasons after the burn, species richness was not significantly different at any time. These fluctuations are attributable to the proportion of plants resprouting vegetatively or regenerating from seed (Purdie 1977a; Purdie 1977b). Some of the shrubs resprouted from seed, which takes longer than vegetative recovery, and some had not returned at all over the period of the study — although this result could be due to difficulties identifying seedlings, e.g. telling apart *Leucopogon collinus* (White Beard-heath) and *L. ericoides* (Pink Beard-heath). In contrast, all of the graminoids resprouted vegetatively, a situation similar to Californian chaparral, where species richness of herbaceous perennials remained constant throughout succession, although cover remained quite low (Keeley *et al.* 1981).

Vegetation floristics and structure, and the presence of litter, bare ground and invertebrates all influence the distribution of small mammals in heathlands (Moro 1991). Generally, the low nutrient soils and high levels of toxic compounds in the leaves make heath species unpalatable for herbivorous mammals (Dwyer *et al.* 1981). However, the change in vegetation cover and availability of green pick obviously makes burnt heathland more favourable (Specht 1981d), as shown by the increase in scats in the present study directly after the burn. For rabbits and some native animals (e.g. wallabies) the green pick after the burn was important in influencing their movements. However, for potoroos and bandicoots the presence of shelter and invertebrates appeared more important (Haseler 1994). This was apparent in the present study by the decrease in animal diggings in the burnt areas for two seasons after the burn.

Combined effects of fire and fertilisers

Due to the lack of response of the heathland to the addition of fertiliser, it is not surprising that there were no interactive effects of fertilisers and fire for any of the variables measured. However, the soil tests for nitrogen and phosphorus show some underlying patterns which may have confused the experimental results. Phosphorus seemed to be retained in the soil over the 3 years of the study, as levels were significantly higher at the end of the experiment than the beginning. Retention of phosphorus after fertilisation has been found by Heddle and Specht (1975), where growth of an indicator species showed that 3.6% of the original superphosphate application was retained in the ecosystem after 22 years. Another paper shows a retention range of 11–71% in a sandy soil (Ozanne *et al.* 1961). Phosphorus concentrations can take approximately two years to decline by 50% in fynbos (Witkowski *et al.* 1990a) and 5–13 years to be lost or fixed into insoluble compounds on abandoned farmland on heathland soil in southern Britain (Pywell *et al.* 1994). Phosphorus is trapped in the surface soil, distributed through deep sand and subsoil and stored in the vegetation itself.

However, levels were not only higher in fertilised quadrats but also in both burnt and unburnt controls. Higher levels in the burnt control quadrats could be explained by the addition of nutrients from a fire, the 'ash-bed' effect (Loneragan and Loneragan 1964; Hester and Hobbs 1992), although in heathlands these raised levels generally return to the pre-fire levels within one year (Siddiqi *et al.* 1976a; Adams *et al.* 1994). Nutrient levels after fire can also be highly variable, due to the uneven distribution of pockets of ash (Siddiqi *et al.* 1976a). It is harder to explain the higher levels of phosphorus over time in the unburnt control, although it could be attributed to measurement error. Another possible reason could be normal variation in soil nutrient levels due to seasonal and climatic variation, and the precise site the soil samples were taken. The nutrient cycling activity of the shrub under which the sample was taken can have a considerable effect on

the nutrient levels measured in heathlands (D. Cheal, School of Botany, University of Melbourne, *pers. comm.* 1996).

Nitrogen levels were generally higher at the beginning of the experiment compared to the end, which could be explained by the volatilisation of nitrogen during the fire. Nitrogen levels were also higher in control quadrats compared with fertilised quadrats (in the top 5 cm at the beginning of the experiment and in the 5–10 cm layer at the end of the experiment). The lower levels in fertilised quadrats at the end of the experiment could indicate the enhanced uptake of nitrogen to maintain a balanced N/P ratio (see below). However, this does not explain why levels were lower in fertilised quadrats at the beginning of the experiment. Hester and Hobbs (1992) found nitrate and ammonium were negatively correlated with numbers of plants, and nitrate was negatively correlated with cover, so it is possible this difference at the beginning of the experiment was due to native shrub density or cover. However, although there was a positive correlation between nitrogen levels and shrub cover abundance at the beginning of the experiment, this correlation was for the 5–10 cm layer not the 0–5 cm layer. This appears to be either another case of either normal variation or measurement error.

In heathlands, fire and fertilisers are thought to have the combined effect of changing the nitrogen and phosphorus ratios in the soil. Fire causes the sudden return of phosphorus to the soil (Loneragan and Loneragan 1964), and volatilises nitrogen which then causes a phosphate imbalance. Nitrogen loss can be as high as 95% (Chapman 1967). This can be partially offset by the growth of many nitrogen-fixing species, which can produce enough nitrogen in the short-term to balance the excess phosphorus (Heddle and Specht 1975). However, the addition of phosphorus fertilisers on top of this causes further imbalances (Ozanne and Specht 1981), leading to the 'phosphorus toxicity' effect causing death of many heath seedlings (Specht 1963), although actual germination may not be affected (Siddiqi *et al.* 1976b). In the present study, the low level of fertilisers used, the underlying phosphorus and nitrogen patterns in the soil and the fact that seedling numbers were not counted explain why any phosphorus toxicity effect was not apparent.

Weed invasion

The exotic grass seeds *Dactylis glomerata* and *Lolium perenne* germinated in the first autumn after they were sown. The fact that they failed to persist is interesting, given that they were provided with fertilisers and there was adequate rainfall. Herbaceous species have usually been found to increase in Australian heathlands after fertilisation, either replacing sedges (Connor and Wilson 1967), or slowly changing the heathland into an herbaceous sward (Specht 1963; Heddle and Specht 1975). In the Netherlands, percentage cover and above-ground biomass of the perennial deciduous grass *Molinia caerula* (Purple

Moor Grass) increased in phosphorus-fertilised plots at the expense of the dwarf shrub *Erica tetralix* (Cross-leaved Heath) (Aerts and Berendse 1988). However the levels of nutrients appear to be important. In a different study (Aerts *et al.* 1990), the two shrubs *Calluna vulgaris* (Heather) and *E. tetralix* out-competed *M. caerula* at lower levels of fertilisers (equivalent to 100 kg/ha super), but at the highest levels (approximately 400 kg/ha) were out-competed by the grass. In the present study, the frequency of the exotic grasses was slightly higher in the higher level of fertiliser treatment.

Low levels of fertilisers are therefore unlikely to provide enough nutrients for exotic grasses to persist. Agricultural trials of annual and perennial grasses on sandy heath-type soils found that they were able to germinate at low levels of nutrients and could persist if competition was low (Riceman 1945 in (Specht 1963), but would be excluded by competition in a vigorously growing heathland. Competition could take the form of shading, lack of water, predation by ants or mammals or a combination of these factors.

Other explanations for the inability of the exotic grasses to become established could be the lack of soil disturbance and/or the presence of native and exotic herbivores. Soil disturbance was not included in the experiment on the grounds that herbaceous species can invade heathland communities without it (Specht 1963; Heddle and Specht 1975) or, if disturbance was still necessary, that the frequent fire history or the presence of herbivores would provide an adequate disturbance factor (Hobbs and Atkins 1988). Increased fire frequency and an increase in nutrient-laden dust proved enough of a disturbance for invasion of exotic grasses in a closed heathland in Western Australia (Bridgewater and Backshall 1981). Native herbivores can also be a significant disturbance factor, digging and burrowing in surface layers of soil (Hobbs 2001). However, even if physical soil disturbance had been included it may not have increased germination, as a study by Hobbs and Atkins (1988) shows that, although disturbance increased establishment success of two exotic weeds, subsequent growth was far higher when disturbance was combined with high nutrient additions (in the order of 250 kg/ha super).

The obvious presence of native and exotic herbivores could be another factor in explaining the absence of exotic grasses. Both the exotic grasses sown are pasture grasses and thus much more palatable than the scleromorphic and siliceous shrubs and graminoids usually found in heathlands. If this were the case, there was not enough feed provided by these grasses to exert a large influence on the movement of the herbivores, as there was no significant differences in scat numbers between sown and not sown quadrats.

Establishment of the exotic grasses was also not aided by fire, reflecting the finding of some researchers that fire does not necessarily cause an increase in exotic species (Hobbs and Atkins 1991; Hester and Hobbs 1992), although others have found that it does enhance weed invasion (Keeley *et al.* 1981; MacDonald *et al.* 1988; Milberg and Lamont 1995). The

combination of fire and soil disturbance can facilitate the invasion by exotics into heathlands (Hobbs and Atkins 1991). The present study did not find that sown exotic grasses established at the expense of native grasses and herbs as found in a Californian chaparral post-fire succession (Keeley *et al.* 1981).

The important issue in these results is not why the exotic grasses did not persist — the environment could have been manipulated in many different ways to favour exotic grasses. One example comes from a dry heathland in Tierra del Fuego, where *Dactylis glomerata* seeds were broadcast over an area that had been ploughed to a depth of 15 cm and fertilised with either nitrogen or phosphorus with lime, the resulting disturbance causing a large drop in shrub cover and high yield responses from the grass (Mendoza *et al.* 1995). Many heathland areas in Tasmania have similarly been cleared, fertilised and sown with grasses, with a massive reduction in native shrub cover (Kirkpatrick 1977; Kirkpatrick and Dickinson 1982; Biodiversity Unit 1995). What is important about the results of the present study is that, with no soil disturbance and low nutrient additions, the grasses *Dactylis glomerata* and *Lolium perenne* were unable to persist in a heathland.

The present study supports the general finding that heathlands can be quite resistant to invasion by exotic species, at least when compared with other native vegetation communities (Hobbs and Atkins 1988; Hester and Hobbs 1992). The exotic herb *Hypochoeris radicata* did not increase due to either burning or fertilising or the two combined. Again, it is likely that higher levels of fertilisers would have achieved an increase in the presence of this weed, particularly as it can be a very successful coloniser (Amor and Stevens 1975) and was found after the addition of fertiliser even though it was not previously present in Specht's (1963) and Heddle and Specht's (1975) heathland studies in South Australia. Other serious weeds, for example *Baccharis halimifolia* (Groundsel Bush) from the coastal lowlands of Queensland, have also become established when high levels of fertilisers were added to a heathland (Connor and Wilson 1967). A return to the site of the present study in 10 or 20 years might show such changes and, if not, would confirm the hypothesis that small additions of phosphorus do not affect the vegetation community.

CONCLUSION

In the short term, low levels of fertilisers do not appear to have an effect on community composition and cover variables in Tasmanian coastal heathland. Burning reduced the live plant cover, which quickly returned to approximately 70% of pre-burn levels. There was no fire-induced increase in species richness, and no interactive effects between fire and fertilisers. It is possible that some underlying soil nitrogen patterns confused the results, or that the experimental period was not long enough for effects to become apparent. A return to the experimental site in 10 or 20 years might show delayed changes and, if not, would

confirm the hypothesis that small additions of phosphorus do not affect the vegetation community. The present study also shows that, with no soil disturbance and low nutrient additions, exotic grasses are unable to persist.

4. Effects of fire and fire-fighting foam

INTRODUCTION

The use of fire-fighting chemicals in Australia is rapidly increasing and becoming more accepted as a necessary part of wildfire suppression and mopping up operations (Adams and Simmons 1999). In Western Australia, purchase of extinguishment additives in 1991/92 was 15% of the fire suppression budget (Rawet *et al.* 1996) and, in Victoria, fire-retardants are used on a regular basis (Christmas Hills Fire Brigade 2000). Fire-suppressant foams are also used regularly in Tasmania. In the 2000/01 fire-fighting season, approximately 2560 litres were purchased by the Parks and Wildlife Service (A. Pyrke, Fire Management Section, Parks & Wildlife Service Tasmania, Department of Primary Industries, Water and Environment, *pers. comm.* 2001), 7040 litres by the Tasmanian Fire Service (M. Chladil, Tasmania Fire Service, *pers. comm.* 2001), and 9600 litres by Forestry Tasmania (T. Geard, Fire Management Branch, Forestry Tasmania, *pers. comm.* 2001). For Forestry Tasmania this represents only 3% of the overall fire-fighting expenditure (T. Geard, *pers. comm.* 2001). In all cases the chemicals purchased are Class A foams.

World-wide, fire-fighting chemicals are used in a range of ecosystems and land tenures, but most often in areas of high wilderness or ecological value (Larson and Newton 1996; Finger *et al.* 1997) and this is also true for Australia (Adams and Simmons 1999). Although concerns were raised about possible environmental effects of these chemicals in the 1970s (Dodge 1970), the focus of research has been on toxicity to humans and fauna and the effects on aquatic ecosystems (for example Gaikowski *et al.* (1996); Hamilton *et al.* (1996); McDonald *et al.* (1996); McDonald *et al.* (1997); Buhl and Hamilton (1998); Buhl and Hamilton (2000)). Although concerns have also been raised about the fertilising affect of fire-fighting chemicals (Gill 1977; Johnson and Sanders 1977), very little detailed research has been conducted on vegetation responses and virtually none on Australian species.

Given the low-nutrient status of heathland soils, the ubiquity of fire and the presence of nutrients in fire-fighting chemicals, Australian heathlands provide a unique opportunity to test the environmental impacts of these chemicals. Therefore, the aim of this chapter is to investigate the effects of a commonly used fire-fighting foam on Tasmanian coastal heathland vegetation. Field experiments were designed to measure the effects of foam on the vegetation community as a whole, and on individual plant flowering and canopy growth and health. The influence of the foam on soil nitrogen and phosphorus is also considered.

UNIVERSITY OF TAS LIBRARY

Fire-fighting chemicals

There are two main types of fire-fighting chemicals, long-term fire retardants and short-term fire retardants. Long-term fire retardants have been used in Australia for over 20 years (Christmas Hills Fire Brigade 2000) and are based on fertiliser salts — typically ammonium phosphate and ammonium sulphate. The retardant effect occurs by chemical modification of the combustion process to form a barrier between the fire and the fuel (Bradstock *et al.* 1987). Their effectiveness remains after the water has completely evaporated and until they are washed away by rain or removed by erosion (USDA 1988). Long-term retardants are usually applied aerially, which makes them particularly useful against fires in remote locations. Accidental applications or spills of retardants have been implicated in fish deaths so their potential adverse affects on aquatic ecosystems are high (Finger 1995; Finger *et al.* 1997; Hamilton *et al.* 1998). In Australia, commonly used retardants are Phos-Chek® D75-F and D75-R, and Fire-Trol® GTS-R (Adams and Simmons 1999).

Short-term fire retardants include both retardants and fire-suppressant foams, and it is the fire-suppressant foams that are the focus of the present study. Both types depend on the water they contain or are mixed with to suppress a fire (USDA 1988). Fire-suppressant foams, also known as Class A foams, generally contain surfactants, and foaming and wetting agents. The surfactants reduce the surface tension of water allowing it to spread further (Schaefer 1996; Vyas *et al.* 1996), while the foaming and wetting agents influence how fast the water drains from the foam and how well the foam adheres to the fuel surfaces (USDA 1988). By using foams, the amount of water required to contain a fire can by reduced by over 60% (Schlobohm 1993). Some formulations also act by slowing the evaporation of water and reducing air contact with the fuel by insulating the fuel source from the heat of the fire (Hamilton *et al.* 1998). Foams can be applied by air but are more commonly added to ground-based portable tanks mounted on trucks or trailers (Christmas Hills Fire Brigade 2000).

In Australia, fire-suppressant foams have been used extensively only in the last few years (Christmas Hills Fire Brigade 2000), but are gradually replacing long-term fire retardants (Adams and Simmons 1999). Commonly used fire-suppressant foams are Ansul Silv-Ex®, Angus ForExpan S®, Fire Quench®, 3M Firebreak® and Phos-Chek® WD-881 (Adams and Simmons 1999). The particular chemical used by a fire-management authority often varies with price or availability, although Angus ForExpan-S is probably the most commonly used foam in Australia (Adams and Simmons 1999).

Effects on vegetation

Research into the effects of both long-term retardants and short-term fire-suppressant foams has been carried out in grasslands and forests in North America, with only one published study in a mixed eucalypt forest in Australia (see Bradstock *et al.* (1987)). Most research investigates either retardants or foams but rarely compares the two. Application of retardants generally results in both a fertilising effect and decrease in species richness, whereas foams have lesser impacts.

The retardant Phos-Chek G75-F was applied to a North Dakota prairie just prior to burning in an experiment by Larson and Newton (1996). The retardant caused an increase in biomass, whether or not the plots were burned, but this effect did not last into the following year. The species most influenced was *Poa pratensis* (Bluegrass), a cool-season grass. Growth of other species was not affected, however, the woody plant *Symphoricarpos occidentalis* (Snowberry) showed an increase in the number of leaves per shoot where retardant was applied (Larson and Newton 1996). The other effect of Phos-Chek G75-F was to depress species richness for a short period.

Similar results were found by Larson and Duncan (1982). Application of a diammonium phosphate retardant to annual grasslands in California resulted in a doubling of biomass, which fell in the following year but still remained higher than control plots. However, there was no effect on species composition or the forb-to-grass ratio.

In Australia, an ammonium sulphate retardant mixture was dropped onto a New South Wales forest of mixed eucalypt and *Angophora costata* (Smooth-barked Angophora) (Bradstock *et al.* 1987). Leaf death occurred within a week of the treatment and continued for several months. However, there was no significant difference from untreated areas 10 months later, and no tree mortality was seen. Understorey shrubs were affected to a greater or lesser degree, depending on the amount of retardant cover. The most severe responses were the death of some individuals of *Dodonaea triquetra* (Large-leaf Hop-bush) and *Acacia longifolia* (Sallow Wattle), and the decrease in cover of 19 of the 45 understorey species one year after treatment (Bradstock *et al.* 1987).

The effects of fire-suppressant foams are more relevant to the current study. The most detailed research is the North Dakota prairie experiment by Larson and Newton (1996), where the foam Silv-Ex was used. Overall, there was little effect from the foam, although there was a small decrease in species richness. Additional effects measured were increased leaf length but decreased shoot growth of *Symphoricarpos occidentalis*, and increased insect herbivory in *S. occidentalis* and *Rosa arkansana* (Prairie Wild Rose) (Larson and Newton 1996).

Other effects of foams vary from none to short-term to more severe. In North America, treatments of 3M FB100 and Fire-Trol FireFoam 103 produced no adverse effects on leaf colour or growth in *Picea glauca* (White Spruce), *Pseudotsuga menziesii* (Douglas Fir), *Thuja plicata* (Western Red Cedar), *Pinus contorta* ssp. *murrayana* (Lodgepole Pine) and *Populus* sp. (Cotton Poplar) tree seedlings (Norecol 1989). Short-term effects were seen in Spain, where applications of Silv-Ex to tree species resulted in initial caustic effects on species which disappeared after four months (Norecol 1989). Short-term effects were also found in a study by MacMillan Bloedal where a variety of foam products were applied over a two-year period. The foams resulted in blight, yellowing and growth retardation in *Tsuga heterophylla* (Hemlock Fir) and *P. menziesii* trees in the first year, which had disappeared by the next spring (Norecol 1989). Both short- and long-term effects from Silv-Ex were found in an experiment conducted by the Canadian Forestry Service. Trees watered with a 1% solution died after two months (Norecol 1989). Small amounts of leaf edge browning were seen in *Populus tremuloides* (Trembling Aspen) leaves but no adverse affects were found in *Populus balsamifera* (Balsam Poplar) or *Betula papyrifera* (White Birch) (Norecol 1989).

In Australia, there are only three studies relevant to fire-fighting foam effects on vegetation. In a glasshouse trial, no effects on the growth characteristics of eight species from the families of Fabaceae, Mimosaceae, Myrtaceae, Poaceae and Proteaceae were recorded from different Class A foam concentrations (Hartskeerl 1999). In a field study, Silv-Ex was found to temporarily depress flowering in *Sprengelia incarnata* (Pink Swamp Heath) in a Tasmanian buttongrass moorland (Marsden-Smedley 1992). In the third, fire-fighting foams were not directly studied, but the presence of surfactants caused a decline in *Araucaria hetrophylla* (Norfolk Island Pine) along Sydney coastal beaches. Detergents discharged into ocean sewage outfalls became incorporated in airborne seaspray and affected the pines by enhancing the effects of salt on the foliage by modifying the waxy coating of the leaves (Moodie *et al.* 1986).

Fire-fighting foams are obviously valuable tools for land managers, but their common usage highlights the need to evaluate their potential ecological impacts. Any decision to use fire-fighting foams involves the careful balancing of factors such as the effectiveness of their suppression of a fire, the cost of fire-fighting equipment, the safety of the fire fighters, the value of the threatened ecosystem or property and the possible long- and short-term effects on terrestrial and aquatic ecosystems of the foam (Schlobohm 1993). In some cases foams will be less damaging than other suppression techniques such as trails made by heavy earth-moving equipment or rake-hoes (Bradstock *et al.* 1987). However, in ecosystems sensitive to disturbance or of a high natural value the potential for changes to the vegetation community may not be acceptable. Adams and Simmons (1999) suggest that

existing guidelines for the use of foams be strictly enforced and their widespread use be minimised until their potential ecological impacts can be more fully assessed.

STUDY SITE

The study site is within the Mt William National Park on the far northeast coast of Tasmania, approximately 100 km northeast of Launceston (Figure 4.1). Mt William National Park was declared in 1973 and enlarged in 1977. It is currently 18,690 ha in size. A part of it had previously been a pastoral property grazed by sheep. Some areas of the park were fertilised and sown to improve the pasture, both when it was a pastoral property and for a few years after it became a national park. The geology of the area is dominated by a granite coastal plain, with pure sand beaches and impoverished soils (Department of Lands Parks and Wildlife 1987). The climate is temperate and maritime, with prevailing westerly winds. The main vegetation is coastal heathlands and eucalypt and banksia woodlands. The altitude and relief are generally very low, i.e. below 100 m, the highest point being Mt William itself at 214 m ASL.

The experimental site itself is within 100 m of the coast in the northern end of the Park, south of Cape Naturaliste. The soils are acid deep leached well-drained sands (never fertilised). The vegetation is a low open heath without a eucalypt overstorey that most resembles a 'Honeysuckle Dry Heath' (Kirkpatrick and Harris 1999b). Common shrub species are *Aotus ericoides* (Golden Pea), *Banksia marginata* (Honeysuckle or Silver Banksia), *Baeckea ramosissima* (Baeckea or Rosy Heath Myrtle) and *Leptospermum scoparium* (Teatree or Manuka). *Xanthorrhoea australis* (Austral Grasstree) is also common. The understorey consists of sedges and herbs, most commonly *Patersonia fragilis* (Short Purple Flag Iris), *Lepidosperma concavum* (Sand-hill Sword Sedge), *Gonocarpus tetragynus* (Common Raspwort) and *Hypolaena fastigiata* (Tassel Rope-rush).

METHODS

There were two different treatments (fire and foam) in four different combinations (Table 4.1). Permanent quadrats were set up in two adjacent areas, one of which was designated to be burnt. The two areas were separated by a single-lane dirt 4WD track and have different fire histories (see below) (Figure 4.1). There were 15 replicates of the four combinations, resulting in a total of 60 quadrats. Thirty quadrats were placed in the area to be burnt (south of the track) and 30 in the unburnt area (north of the track), and the foam treatment was randomly allocated to 15 of each.

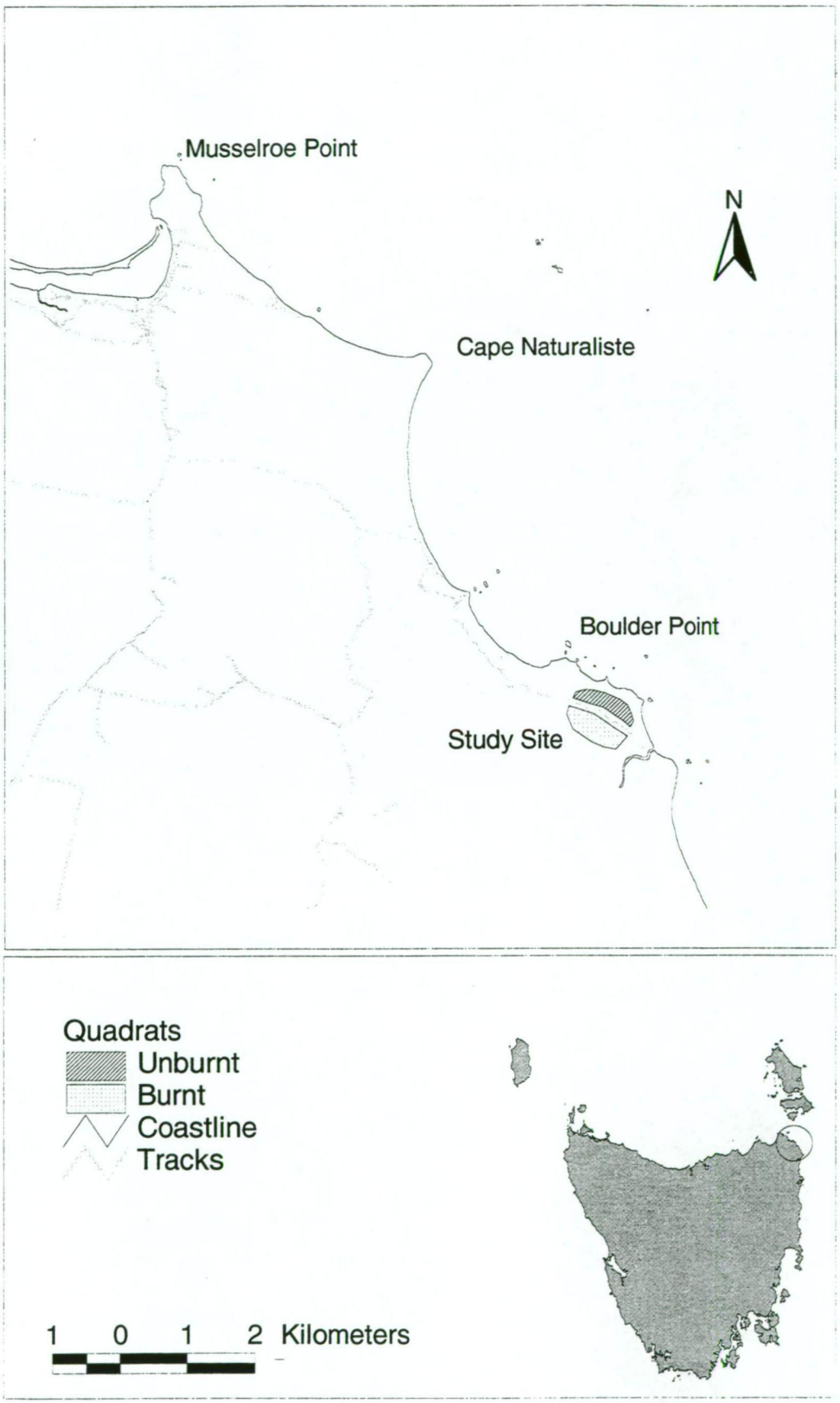


Figure 4.1. Map of Mt William National Park in northeast Tasmania, showing the study area and position of burnt and unburnt quadrats

Table 4.1. Mt William experimental treatment combinations

Treatment	Code
FIRE + FOAM	BF
FIRE	B
FOAM	F
CONTROL	C

Fire treatment

The fire treatment south of the 4WD track was done as the pre-burn part of a fuel reduction and habitat/biodiversity maintenance burn by the Parks and Wildlife fire crew, scheduled for approximately 2000 ha of Mt William National Park. Control burns are usually done by Parks and Wildlife in spring and autumn, and the availability of the fire crew and the wait for safe weather conditions led to the burn being conducted in late autumn. There had been 32 mm of rain in the week prior to the burn and it had been 4 days since the last rain. The lighting method was by hand-held drip torch. The burn took place on 12 May 1997, starting at 3.30 p.m. The maximum temperature was 14°C, the relative humidity was 62% and the wind speed at 1–7 m was 7 km/h coming from the northnorthwest. The fire front progressed at 4.5 m/min with a flame height of 2 m. The experimental site was approximately 90% burnt, with all quadrats thoroughly burnt.

The fire history of the site shows that the southern side of the track (burnt in this experiment) had been burnt in 1987/88. There is no record for the last fire on northern side of the track, but the greater size of *Banksia marginata* bushes indicates a longer time since fire.

Foam treatment

The foam chosen was ForExpan-S produced by Angus Fire Armour Pty Ltd, commonly used by the Tasmanian Parks and Wildlife fire crew. ForExpan–S functions as a surfactant and is a proprietary mixture of ammonium deceth 2,2 sulfate, 2(2-butoxyethoxy) ethanol, ethanol, sodium myriteth 3 sulfate, myriteth-3, and 1-dodecanol (Angus Fire Armour 1997; Hamilton *et al.* 1998). When a sample of the foam used in this experiment was sent to the Tasmanian Government Analytical and Forensic Laboratory for analysis of the major nutrients, explosions of the product meant total phosphorus and total nitrogen were unable to be assessed. An estimate of the nutrient levels are shown in Table 4.2. ForExpan–S is supplied by the manufacturers as a liquid concentrate that is prepared for use in the field by mixing 0.1–1% of concentrate with water (Angus Fire Armour 1997; Hamilton *et al.* 1998). This mixture, when aerated, produces large quantities of foam.

Table 4.2. Approximate levels of phosphorus, nitrogen and pH in ForExpan-S foam

nutrient	mg/L
dissolved reactive phosphorus	140
ammonia	210
reaction (pH)	7.2

For the present study, the foam was added to dam water in the water tank of a 'slipon' fire-fighting unit at a rate of 1% foam to water. This amount, being the upper limit of the recommended amount, was chosen to maximise any potential effects on the vegetation. The foam treatment was applied in the four days directly after the burn. This time was chosen for foaming in order to simulate mopping up operations usually done to ensure fire boundaries are safe post-burn. The foaming could not be done just prior to or during the burn as the quadrats would then not have burnt. For the first day of the foam treatment, a six-wheel tanker was used, with a 1700-litre tank that could treat 6 quadrats without refilling. For the subsequent three days, a smaller slipon was used with a 460-litre tank that could only cover 2 quadrats. The approximate amounts used are shown in Table 4.3, the small amounts of nutrients in the foam resulting in very small amounts when calculated as kg/ha. The foam and water mixture was applied with a hose until a sixth (for the first day) or a half (for subsequent days) of the tank had been used each time (Figure 4.2a and b). Equal amounts were applied to burnt and unburnt quadrats.

Table 4.3. Amounts of foam and water and approximate amounts of phosphorus and ammonia applied per 5 x 5 m quadrat

	foam ($l/25m^2$)	water ($l/25m^2$)	phosphorus ($mg/25m^2$)	(kg/ha)	ammonia ($mg/25m^2$)	(kg/ha)
first 6 quadrats	2	200	280	0.112	410	0.164
All other quadrats	2	230	280	0.112	410	0.164

Field measurements

Vegetation community

The quadrat size was 5 m x 5 m, set out along either side of the fire-boundary track at approximately 30 m from the track. The distance between quadrats was never less than 10 m or more than 35 m and was chosen using a random number table. Quadrats were marked using 20 cm long metal deck spikes at each corner and a metal star picket placed at every third quadrat. Quadrat numbers were stamped on stainless steel metal tags that would not melt in the fire. The position of each star picket and a few other quadrats was recorded with a hand-held GPS (Magellan ProMARK X) from Parks and Wildlife.

Figure 4.2. a) Application of foam treatment to unburnt quadrat

Figure 4.2. b) application of foam treatment to burnt quadrat

The first set of baseline measurements was taken over three field trips in April and early May 1997 (Autumn 1). The quadrats were then visited one month after the burn, in mid-June (Winter 1). Subsequent measurements were taken once each season over the next 12 months, and occurred in September 1997 (Spring), January 1998 (Summer), April 1998 (Autumn 2) and August 1998 (Winter 2).

All vascular plant species were recorded and given a cover value using a modified Braun–Blanquet scale (1, <1%; 2, 1–5%; 3, 6–25%; 4, 26–50%; 5, 51–75%; 6, > 75%). The dominant and subdominant species were recorded for the shrub strata. Species nomenclature follows that of Buchanan (1999), and a full species list is recorded in Appendix 3.

Individual plants

Five common species were chosen to represent different plant families. They were: *Aotus ericoides* (Golden Pea), Fabaceae; *Banksia marginata* (Honey Myrtle or Silver Banksia), Proteaceae; *Leptospermum scoparium* (Teatree or Manuka), Myrtaceae; *Leucopogon collinus* (White Beard Heath), Epacridaceae; and *Xanthorrhoea australis* (Austral Grasstree), Xanthorrhoeaceae. Three individuals of each species in each quadrat were tagged with plasticised metal twist ties. In most of the quadrats it was possible to tag the maximum 15 individuals, but not all quadrats had all 5 species or all 3 individuals of each species. Individual plants were tagged during the baseline score for the unburnt quadrats. There was no baseline score for the burnt quadrats because the plant itself and the identifying tag would have been burnt away. The first measurement for the burnt treatment was therefore not until the summer after the burn when the resprout stems were tall enough to be identified and tagged.

For each individual plant, canopy height and width were measured to the nearest centimetre at the highest and widest point, respectively. The two values were multiplied together to give the canopy area. This gave an estimate of the growth of the canopy. The number of inflorescences and number of flowers in each inflorescence were counted when the plant was flowering. For *Xanthorrhoea*, the height and width of the flowering spike was measured instead.

Canopy health was assessed by a set of four scores on a scale of 1 to 6 (1 being dead or very sick, 6 being most healthy). The first score was for the amount of leaves in the canopy, which gives some indication of the general 'sparseness' or 'fullness' of the canopy. The next two scores were related to colour, firstly the amount of yellow discolouration (chlorosis) and next the amount of red discolouration. This is an assessment of the 'health' of the plant and can indicate nutrient imbalances. Colour was assessed subjectively, because the use of Munsell leaf colour charts proved too time consuming. The fourth score was the amount of attached brown leaves, equivalent to the percentage of dead leaves in the

canopy. This score measures the amount of leaf death that might occur before all leaves are lost . These four scores thus show whether the canopy is lush and green and growing vigorously or, if not, what level and type of stress the plant is experiencing.

Animal scats

All quadrats were cleared of animal scats in the Spring score. For each score after that (Summer 1, Autumn 2 and Winter 2) the number of animal scats deposited since the previous score were counted. Two 80 cm x 5 m quadrats along parallel sides within the vegetation quadrat were used. The scats were identified as either rabbit, wallaby or wombat and were counted and removed.

Analysis

The vegetation community was investigated for patterns with ordinations using Non-metric Multidimensional Scaling (NMS). The purpose of the investigation was to see if there were floristic differences between the burnt and unburnt quadrats prior to treatment, and between the foamed and unfoamed quadrats within each burnt and unburnt treatment. The Multi-Response Permutation Procedure (MRPP) was used to test for multivariate difference between pre-defined groups using the software PC–ORD (version 4). This procedure is a non-parametric version of discriminant analysis, and provides the statistics A (within-group homogeneity) and p (the probability that the observed within-group homogeneity is smaller or equal to that expected by chance) (McCune and Mefford 1999).

One-way ANOVAs were used to test for differences in total species richness and lifeform species richness between foamed and unfoamed treatments, using the statistical package MINITAB (Release 12.2). The lifeform groups were ferns, grasses, graminoids (monocotyledons other than grasses or orchids), herbs, orchids, parasites and shrubs. Burnt and unburnt quadrats were tested separately. The Anderson-Darling statistic was used to test the residuals for normality for each ANOVA. Data were transformed where appropriate, using natural log, square root or arcsine transformations, or the lambda value suggested by a Box-Cox transformation, depending on the type and shape of the data (Sokal and Rohlf 1995). The Box-Cox transformation estimated a lambda value that minimised the standard deviation of the transformed variable. The transformation with the most normal residual was then used. Some variables were not normal even after repeated attempts at transformation and for these the Kruskal-Wallis rank order test for medians was used as a non-parametric alternative to ANOVA.

Individual plant growth was tested using one-way ANOVAs using the methods just described. The differences in canopy height, width and area (height multiplied by width) between foamed and unfoamed treatments were tested separately for burnt and unburnt

quadrats. To tackle the lack of baseline data for the burnt quadrats, and pre-treatment differences found in the unburnt quadrats, the change in growth from one season to the next was also tested. This was calculated by subtracting one season from another and then testing this difference. Where enough flowering occurred, one-way ANOVAs were also used to test each species for significant differences in the number of inflorescences, number of flowers per inflorescence and total number of flowers (number of inflorescences multiplied by the number of flowers per inflorescence) between foamed and unfoamed treatments. Burnt and unburnt quadrats were treated separately. Chi-squared analysis for contingency tables was used to test for differences in the canopy health and leaf colour scores between foamed and unfoamed treatments.

The numbers of animal scats were also tested using one-way ANOVAs or the Kruskal-Wallis rank order test. The data were tested in two ways, firstly, between all four treatment combinations and, secondly, between foamed and unfoamed for the burnt quadrats only. There were not enough scats to test for differences between foamed and control in the unburnt quadrats.

RESULTS

Vegetation community

Ordination of all baseline quadrats showed a marked difference in the floristics of the unburnt and to-be-burnt areas (Figure 4.3a–c). The MRPP result confirmed this separation, showing that the groups based on future treatments were significantly different (Table 4.4). After the treatments had been applied, quadrats continued to separate into clusters on the ordinations based on whether they were burnt or unburnt, and this is validated by the MRPP results (Table 4.4). This meant that the burnt and unburnt areas had to be treated as two separate experiments. All post-burn ordinations look similar to the pre-burn ordination and so are not reproduced here.

When the burnt and unburnt quadrats were ordinated separately and tested for differences between foamed and unfoamed treatments, there were no significant floristic differences for either the baseline or the post-treatment scores (Table 4.4).

Total species richness was then compared between foamed and unfoamed quadrats (Figure 4.4). In the burnt treatment, richness was significantly lower in burnt and foamed quadrats compared with burnt only quadrats in the Spring directly after the burn (Table 4.5). There were no significant differences in any other season or in the unburnt treatment.

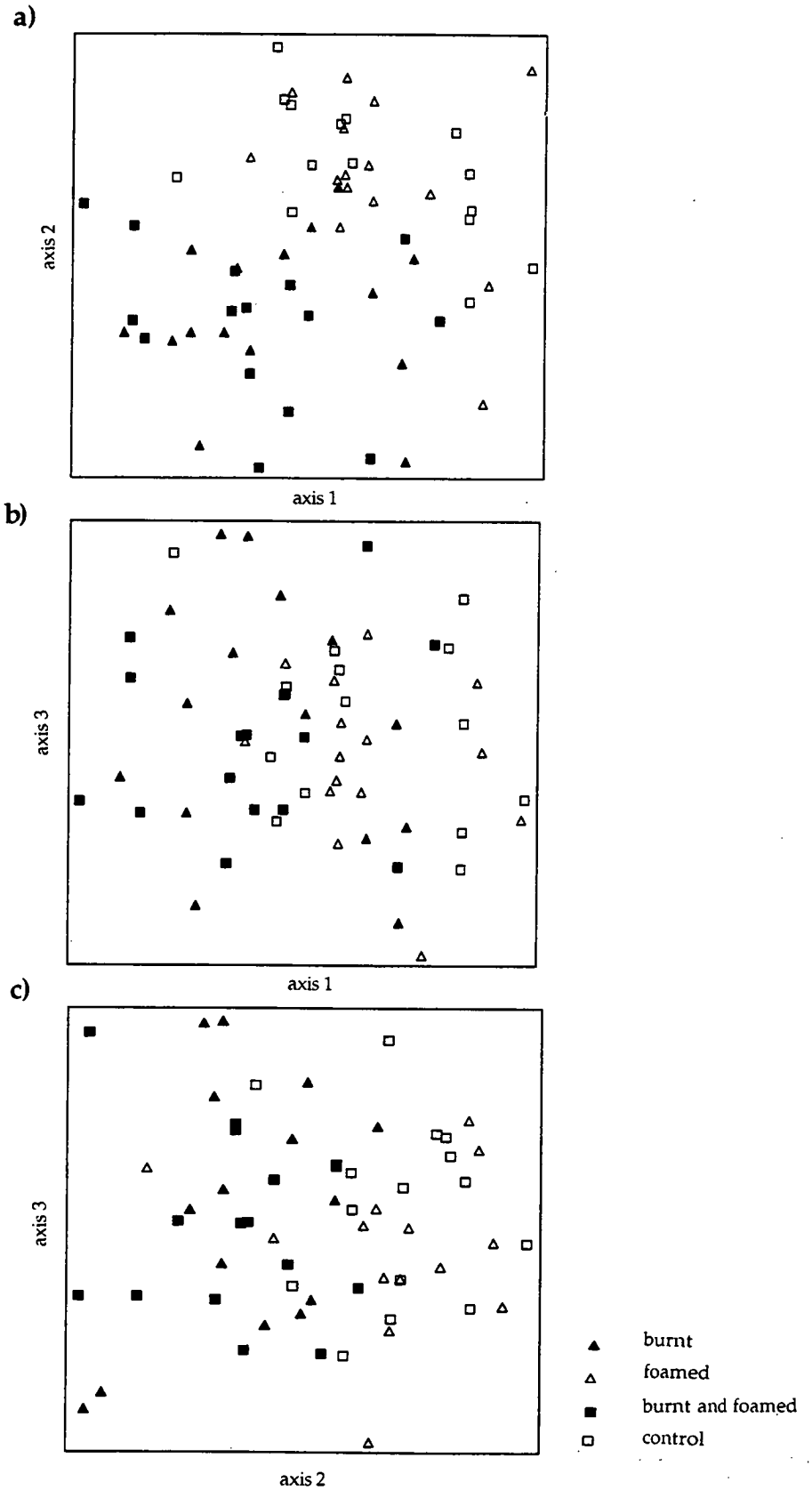


Figure 4.3. Three-dimensional NMS ordination of quadrats before burn treatment a) axes 1 x 2, b) axes 1 x 3, c) axes 2 x 3

Table 4.4. Ordination results showing final number of dimensions, stress, and A and P values for MRPP

ALL PLOTS	dimensions	stress	A	P	signif
autumn 1	3	16.02098	0.04664128	0.00000835	***
spring	1	8.66832	0.51941855	0.00000000	***
summer	3	11.14395	0.18186976	0.00000000	***
autumn 2	3	13.68064	0.09663768	0.00000003	***
winter 2	3	12.56170	0.10183515	0.00000000	***
BURNT					
autumn 1	3	12.12763	-0.0048898	0.67507557	ns
spring	2	23.09453	0.00645003	0.29398155	ns
summer	2	11.04076	-0.0060558	0.51534146	ns
autumn 2	3	10.51450	-0.0032614	0.51947199	ns
winter 2	2	17.43327	0.0016285	0.33961314	ns
UNBURNT					
autumn 1	2	20.18925	-0.0082586	0.83044576	ns
spring	3	11.78037	0.00702673	0.29079868	ns
summer	2	18.71781	-0.0024646	0.48443816	ns
autumn 2	3	11.87239	-0.0116579	0.94352445	ns
winter 2	3	10.27342	-0.0092409	0.81714480	ns

ns, not significant; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

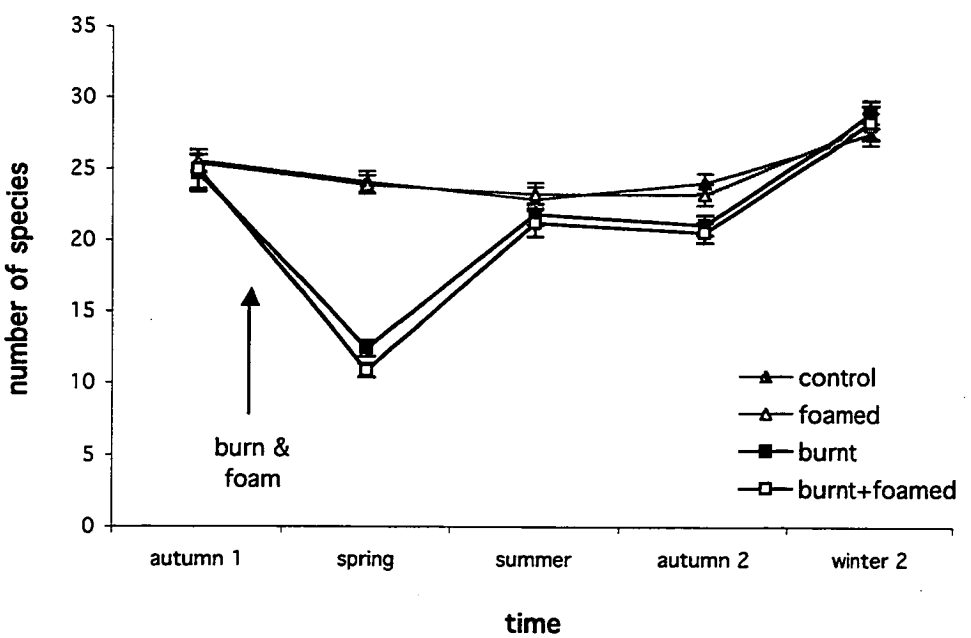


Figure 4.4. Average total species richness over time. Standard error bars are shown. Arrow shows timing of burn and foam treatments

Table 4.5. Significant results for one-way ANOVAs and Kruskal-Wallis rank order test of species richness and lifeform species richness. Figures show average species richness followed by standard error in brackets

variable	season	B	BF	P		C	F	P	
total	Autumn 1	24.67 (1.27)	24.93 (1.36)	0.887	ns	25.53 (0.76)	25.33 (0.64)	0.843	ns
	Spring	12.40 (0.59)	10.87 (0.48)	0.046	*	24.07 (0.73)	23.87 (0.64)	0.839	ns
	Summer	21.80 (0.76)	21.20 (0.99)	0.633	ns	22.87 (0.83)	23.20 (0.82)	0.777	ns
	Autumn 2	21.07 (0.70)	20.53 (0.71)	0.597	ns	24.00 (0.69)	23.20 (0.73)	0.430	ns
	Winter 2	28.87 (0.96)	28.27 (1.21)	0.701	ns	27.47 (0.77)	28.07 (0.92)	0.620	ns
fern	Autumn 1	2.33 (0.16)	1.73 (0.18)	0.021[1]	*	0.73 (0.23)	0.40 (0.19)	0.200[1]	ns
	Spring	1.67 (0.16)	1.20 (0.11)	0.011[1]	*	0.53 (0.19)	0.33 (0.23)	0.182[1]	ns
	Summer	1.73 (0.12)	1.80 (0.11)	0.671[1]	ns	0.47 (0.19)	0.60 (0.19)	0.537[1]	ns
	Autumn 2	1.73 (0.12)	1.60(0.19)	0.830[1]	ns	0.47 (0.19)	0.27 (0.18)	0.247[1]	ns
	Winter 2	1.73 (0.12)	1.67 (0.16)	0.914[1]	ns	0.47 (0.19)	0.40 (0.19)	0.738[1]	ns
grass	Autumn 1	0.13 (0.13)	0.27 (0.15)	0.326[1]	ns	0.20 (0.11)	0.13 (0.09)	0.630[1]	ns
	Spring	0.13 (0.09)	0.13 (0.09)	1.000[1]	ns	0.13 (0.09)	0 (0)	0.150[1]	ns
	Summer	0.27 (0.12)	0.13 (0.09)	0.369[1]	ns	0.27 (0.12)	0 (0)	0.032[1]	*
	Autumn 2	0.13 (0.09)	0.53 (0.13)	0.022[1]	*	0.40 (0.16)	0.13 (0.09)	0.185[1]	ns
	Winter 2	1.27 (0.25)	1.67 (0.19)	0.208	ns	0.33 (0.21)	0.60 (0.16)	0.092[1]	ns

[1] result from Kruskal-Wallis rank order test. ns, not significant; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

When the species richness for each lifeform was tested separately, foaming affected grasses differently depending on whether they were burnt or unburnt. In the burnt treatment, there were more grass species in Autumn 2 in foamed quadrats. However, in the unburnt quadrats, there were less (i.e. none) grass species in Summer when foamed.

Fern richness was significantly lower in foamed and burnt quadrats compared with burnt only quadrats in Spring. However, this result cannot be attributed to the foaming treatment because fern richness was already lower in the baseline score in Autumn 1 in the foamed and burnt quadrats, pre-treatment.

Very few exotic species were found during the experimental period. Only one grass species, *Aira caryophyllea* (Silvery Hair Grass), was found in one burnt and foamed quadrat in summer. The exotic herbs *Hypochoeris radicata* (Cat's Ear) and *Cirsium* spp. (Thistles) were only present in Winter 2, in two or three quadrats in each of the treatments.

Individual plants

Aotus ericoides

In general, foaming increased growth (canopy height, width and area) in both the burnt and unburnt treatments (Figure 4.5). However, in the unburnt treatment, the height, width and area were already greater in the baseline to-be-foamed quadrats, although only height was significantly so. Therefore, although height was significantly greater in

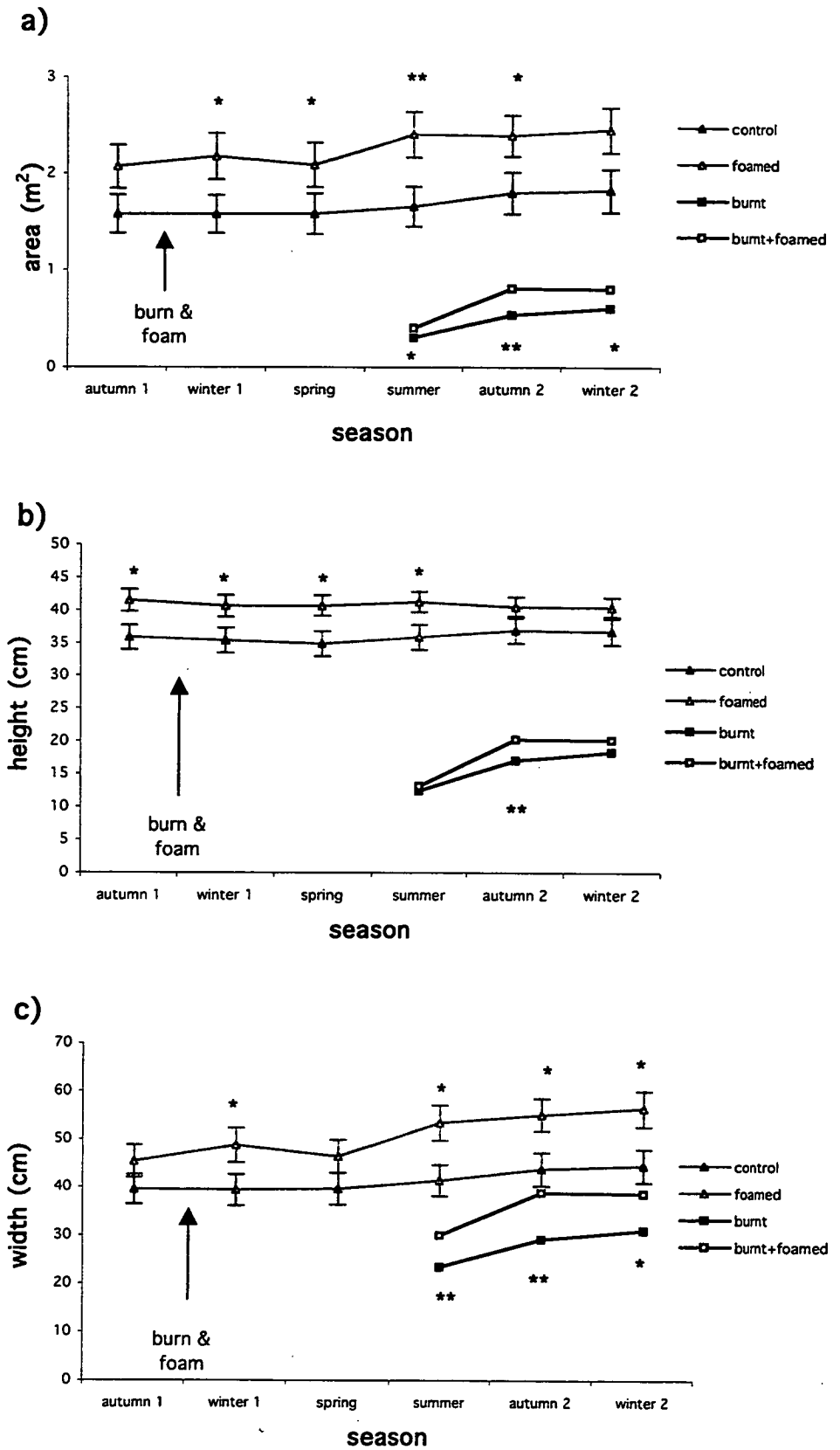


Figure 4.5. *Aotus ericoides* canopy a) height, b) width and c) area over time for burnt and unburnt treatments. Standard error bars are shown (note error bars are too small to be visible on burnt series). Arrow shows timing of burn and foam treatments. *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

foamed quadrats in Winter 1 (one month after foaming), Spring and Summer, these differences cannot be attributed to a treatment effect. Width and area were not significantly different between treatments in the baseline score, but were significantly greater in foamed quadrats in Winter 1, Summer and Autumn 2. In Spring just the canopy area was greater and in Winter 2 just the width was greater in the foamed quadrats. From Spring to Summer, the canopy width and area increased significantly for foamed quadrats but stayed the same for unfoamed quadrats. From Winter 1 to Spring the canopy height decreased significantly in unfoamed quadrats but increased in foamed quadrats (Table 4.6).

Table 4.6. Results for Kruskal-Wallis rank order tests for the change in *Aotus ericoides* canopy height, width and area from season to season. Figures show mean values for height and width in cm and for area in m^2, followed by standard error in brackets

AREA	F	C	P		BF	B	P	
aut1-wint1	0.107 (0.085)	-0.001 (0.080)	0.578	ns				
wint1-spr2	-0.091 (0.069)	0.003 (0.100)	0.802	ns				
spr2-sum2	0.314 (0.092)	-0.004 (0.077)	0.006	**				
sum2-aut2	-0.009 (0.097)	0.139 (0.075)	0.546	ns	0.359 (0.039)	0.201 (0.043)	0.007	**
aut2-wint2	0.058 (0.098)	0.034 (0.055)	0.837	ns	0.017 (0.023)	0.067 (0.033)	0.224	ns
HEIGHT	**F**	**C**	**P**		**BF**	**B**	**P**	
aut1-wint1	-0.822 (0.430)	-0.500 (0.597)	0.148	ns				
wint1-spr2	0.044 (0.347)	-0.452 (1.089)	0.043	*				
spr2-sum2	0.578 (0.532)	-0.762 (0.908)	0.983	ns				
sum2-aut2	-0.733 (0.872)	1.025 (0.732)	0.187	ns	6.028 (0.754)	3.618 (0.972)	0.011	*
aut2-wint2	-0.111 (0.676)	0.316 (0.461)	0.292	ns	0.472 (0.649)	1.515 (0.830)	0.114	ns
WIDTH	**F**	**C**	**P**		**BF**	**B**	**P**	
aut1-wint1	3.289 (1.908)	-0.143 (1.407)	0.222	ns				
wint1-spr2	-2.289 (1.583)	0.286 (1.825)	0.345	ns				
spr2-sum2	7.022 (1.599)	-0.214 (1.794)	0.002	**				
sum2-aut2	1.600 (1.421)	2.214 (1.311)	0.818	ns	6.72 (1.38)	4.06 (1.84)	0.076	ns
aut2-wint2	1.356 (1.588)	0.561 (1.063)	0.549	ns	0.92 (0.71)	1.42 (1.5)	0.701	ns

ns, not significant; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

In the burnt treatment, coppice resprouts from lignotubers were measured. The canopy area and width were always higher in the burnt and foamed quadrats compared with burnt only quadrats, in Summer, Autumn 2 and Winter 2. Height was also greater in foamed quadrats in Autumn 2 (Figure 4.5). When testing the change in canopy growth over time, height and area increased more for foamed and burnt quadrats than it did for burnt only quadrats over Summer to Autumn 2 (Table 4.6).

There were no significant differences in any of the canopy health counts in the baseline score. Leaf yellowing varied with foaming in the unburnt treatment, although in different ways at different times (Table 4.7). In the Winter 1 score one month after the treatments were applied, there were less yellow leaves in the canopy for foamed quadrats. However, two season later in Autumn 2, there were more yellow leaves in the foamed quadrats. For

the burnt quadrats, the scores for canopy fullness and the amount of brown leaves were either all the same or there were too few low scores to test. There were no differences in yellow or red leaf colour.

Table 4.7. Significant results for chi-squared tests for *Aotus ericoides* canopy health

		CANOPY			YELLOWNESS			REDNESS			BROWNNESS		
season	n	chi	DF	P	chi	DF	P	chi	DF	P	chi	DF	P
UNBURNT													
winter 1	87				5.18	1	0.023						
							F < C						
autumn 2	84				9.48	1	0.002						
							F > C						

Banksia marginata

Banksia marginata individuals varied greatly in size, from mature shrubs over 2 m to small shrubs under 50 cm. This resulted in large standard error rates for canopy size and change in canopy size over time. Generally, the growth of mature *B. marginata* individuals and new coppice resprouts was not affected by foaming — there were no significant differences in canopy height, width or area in any season in either the burnt or unburnt treatments (Figure 4.6). However, plants in unfoamed quadrats increased significantly in height from Spring to Summer, whereas those in foamed quadrats stayed approximately the same (Table 4.8). There were no significant differences in the number of flowers between foamed and unfoamed plants.

Foaming had its most dramatic and lasting effect on *B. marginata* leaves, mostly in the unburnt treatment (Table 4.9). In Spring in the unburnt treatment, plants in foamed quadrats were significantly more reddened, showed more yellowing, and there were more brown dead leaves in the canopy compared with unfoamed quadrats. In Summer, there were also more reddened and brown dead leaves in foamed quadrats. In the next season, Autumn 2, there were only more brown dead leaves. In Winter 2, the final season, plants in foamed quadrats again showed more yellowing. Yellow and red discolouration usually consisted of spots of various sizes all over the leaf surface. In summer, some growing tips were dark red rather than the usual brownish-green.

In the burnt treatment, the canopy height and area of plants in burnt quadrats increased significantly more than those of plants in burnt and foamed quadrats, from Spring to Summer (Table 4.8). The only other significant difference between foamed and unfoamed treatments was a less full canopy in plants in the foamed quadrats in Autumn 2 (Table 4.9).

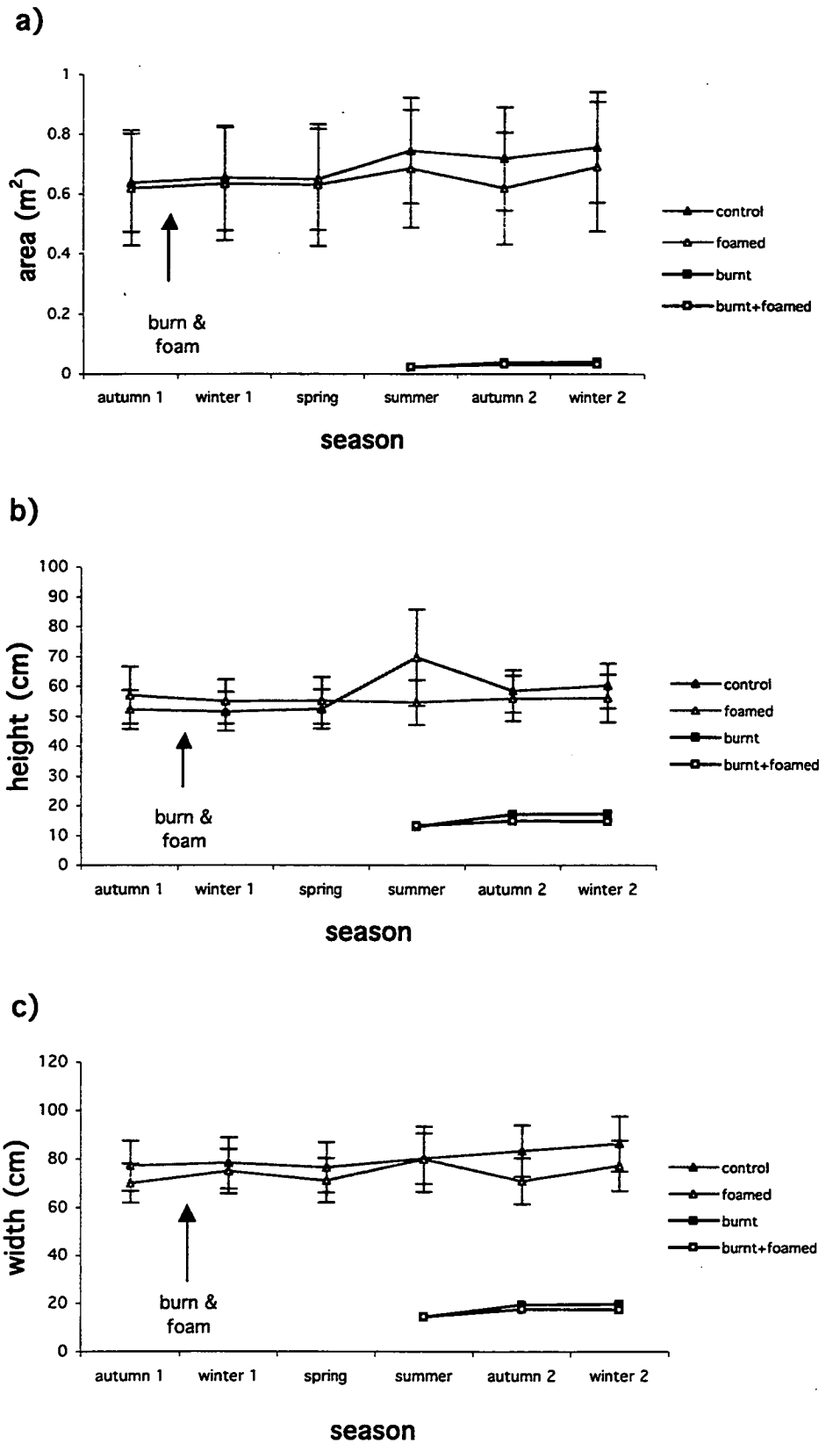


Figure 4.6. *Banksia marginata* canopy a) height, b) width and c) area over time for burnt and unburnt treatments. Standard error bars are shown (note error bars are too small to be visible on burnt series). Arrow shows timing of burn and foam treatments

Table 4.8. Results for Kruskal-Wallis rank order tests for the change in *Banksia marginata* canopy height, width and area from season to season. Figures show mean values for height and width in cm and for area in m^2, followed by standard error in brackets

AREA	F	C	P		BF	B	P	
aut1-wint1	0.014 (0.036)	0.016 (0.016)	0.162	ns				
wint1-spr2	-0.005 (0.028)	-0.004 (0.025)	0.345	ns				
spr2-sum2	0.054 (0.079)	0.081 (0.075)	0.133	ns				
sum2-aut2	-0.096 (0.089)	-0.079 (0.069)	0.902	ns	0.009 (0.004)	0.017 (0.003)	0.024	*
aut2-wint2	0.070 (0.051)	0.036 (0.033)	0.571	ns	-0.001 (0.002)	0.001 (0.001)	0.328	ns
HEIGHT	**F**	**C**	**P**		**BF**	**B**	**P**	
aut1-wint1	-2.075 (2.343)	-0.610 (2.524)	0.381	ns				ns
wint1-spr2	0.275 (0.406)	0.829 (0.585)	0.105	ns				ns
spr2-sum2	-0.525 (14.660)	16.000 (0.879)	0.037	*				ns
sum2-aut2	-1.500 (14.689)	-15.561 (1.207)	0.794	ns	1.645 (0.819)	4.303 (0.803)	0.005	**
aut2-wint2	0.025 (1.143)	1.732 (0.791)	0.483	ns	-0.548 (0.347)	0.212 (0.625)	0.595	ns
WIDTH	**F**	**C**	**P**		**BF**	**B**	**P**	
aut1-wint1	4.825 (2.293)	1.098 (4.163)	0.163	ns				ns
wint1-spr2	-3.700 (2.487)	-1.756 (2.580)	0.616	ns				ns
spr2-sum2	8.675 (8.513)	1.756 (2.940)	0.427	ns				ns
sum2-aut2	-12.50 (8.723)	-2.976 (3.593)	0.526	ns	3.226 (0.730)	5.333 (1.052)	0.184	ns
aut2-wint2	6.000 (2.500)	2.805 (3.094)	0.347	ns	-0.710 (0.835)	0.273 (0.273)	0.660	ns

ns, not significant; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

There were no significant differences in the number of inflorescences between foamed and unfoamed quadrats in the unburnt treatment (no flowers were present in the burnt treatment).

Table 4.9. Significant results for chi-squared tests for *Banksia marginata* canopy health

		CANOPY			YELLOWNESS			REDNESS			BROWNNESS		
season	n	chi	DF	P	chi	DF	P	chi	DF	P	chi	DF	P
UNBURNT													
spring	81				25.91	1	0.000	31.15	2	0.000	12.74	2	0.002
							F > C			F > C			F > C
summer	81							10.69	2	0.005	9.36	1	0.002
										F > C			F > C
autumn 2	76										4.65	1	0.031
													F > C
winter 2	76				7.66	1	0.006						
							F > C						
BURNT													
autumn 2	64	4.17	1	0.041									
				BF<B									

Leptospermum scoparium

Compared with *A. ericoïdes*, foaming had a lesser and opposite effect on canopy size of *L. scoparium* coppice resprouts. In the unburnt treatment, width in Winter 1, width in Spring, area and width in Summer and area and width in Winter 2 were all lower in the foamed quadrats (Figure 4.7). There were no differences in the baseline score. There were also no significant differences in the burnt treatment. There were no significant differences between the treatments when the change in canopy growth over time was tested.

Autumn 2 was the only time there were enough flowers to test for differences between foamed and unfoamed quadrats in the unburnt treatment. There were less inflorescences and less flowers per inflorescence when foamed, leading to a decrease in the total number of flowers (Table 4.10).

Table 4.10. Significant results for one-way ANOVAs for *Leptospermum scoparium* flowering. INF, number of inflorescences; FLW/INF, number of flowers per inflorescence; FLOWERS, total number of flowers. Figures show mean values, followed by standard error in brackets

	F	C	P	signif
INF	5.94 (1.50)	22.00 (6.56)	0.021	*
FLW/INF	2.94 (0.65)	4.56 (0.60)	0.032	*
FLOWERS	29.40 (13.00)	132.00 (41.30)	0.017	*

ns, not significant; *, $P < 0.05$

Foaming only had one effect on canopy health scores, which was a slightly increased yellowing in control quadrats in the unburnt treatment in Spring (Table 4.11).

Table 4.11. Significant results for chi-squared tests for *Leptospermum scoparium* canopy health

		CANOPY			YELLOWNESS			REDNESS			BROWNNESS		
season	n	chi	DF	P	chi	DF	P	chi	DF	P	chi	DF	P
UNBURNT													
spring	57				3.93	1	0.047						
							F > C						

Leucopogon collinus

L. collinus was the only species studied that regenerates only from seed after fire rather than resprouting. The effect of foaming on seedling canopy size was either very minor or exhibited a delayed response. The only difference was in the unburnt treatment in the two last seasons, where canopy width was smaller in the foamed quadrats (Figure 4.8). Two observations indicate that these results are not conclusive, firstly, that *L. collinus* plants

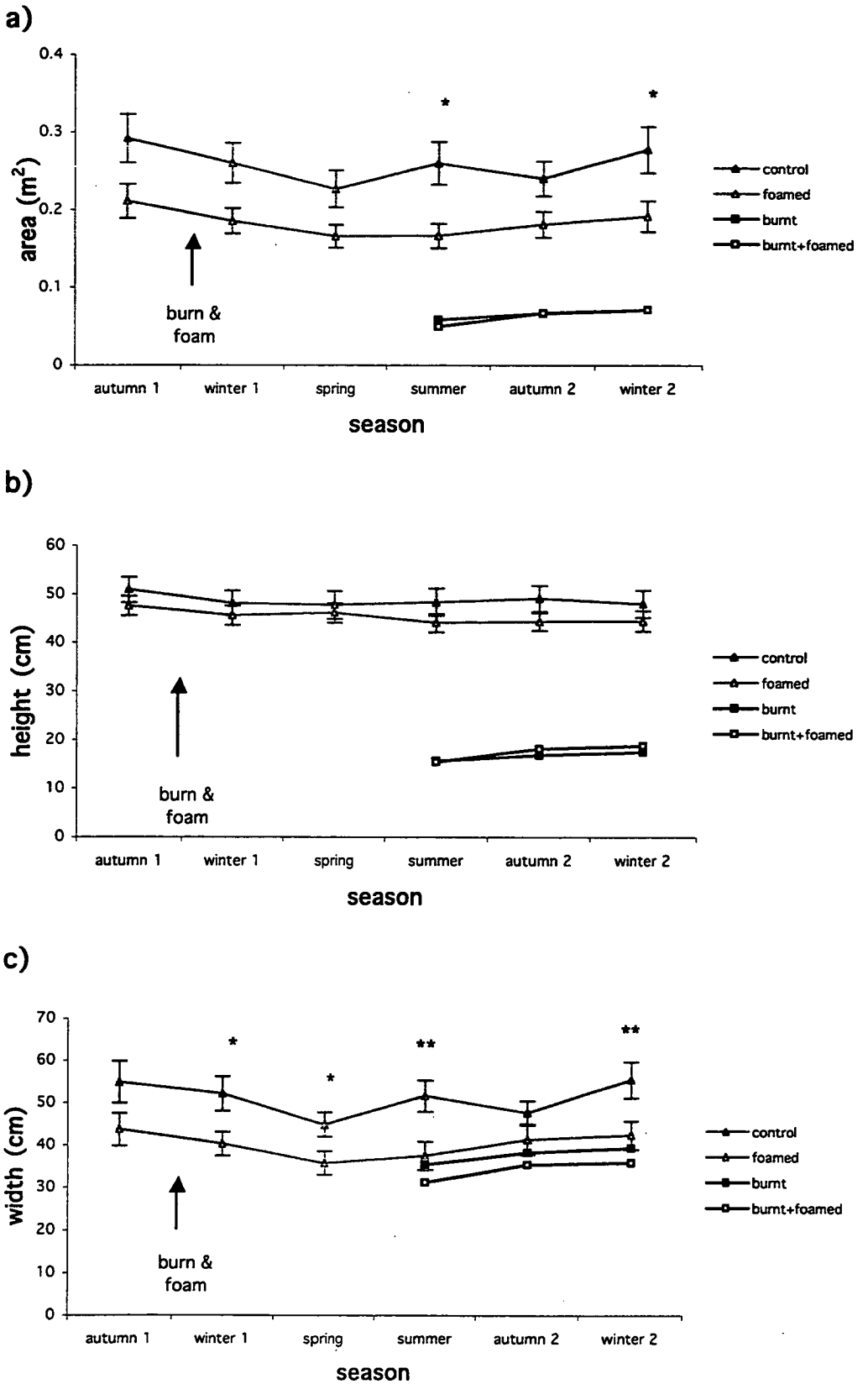


Figure 4.7. *Leptospermum scoparium* canopy a) height, b) width and c) area over time for burnt and unburnt treatments. Standard error bars are shown (note error bars are too small to be visible on burnt series). Arrow shows timing of burn and foam treatments. *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

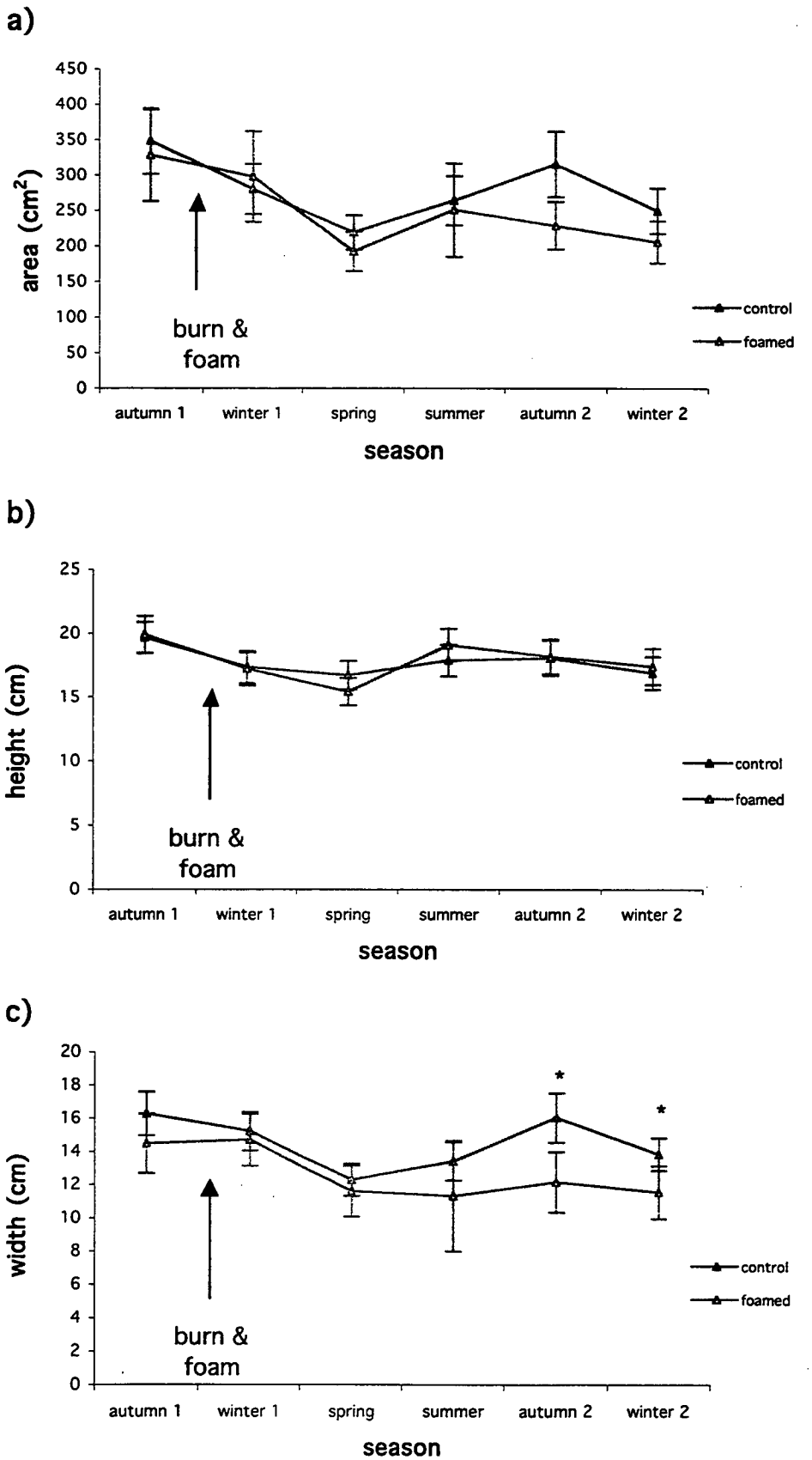


Figure 4.8. *Leucopogon collinus* canopy a) height, b) width and c) area over time for unburnt treatment. Standard error bars are shown. Arrow shows timing of burn and foam treatments. *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

were very small and usually had only a single stem meaning measurement errors, particularly the width, would have more influence on the results and, secondly, that the results were significant only to the 0.05 level. There were no significant differences between the treatments when the change in canopy growth over time was tested. There were no results for the burnt treatment as no seedlings were large enough to test.

Canopy fullness varied in the baseline score — plants in foamed quadrats had less full canopies than plants in control quadrats — but not in any season after the treatments were applied (Table 4.12). Less brown leaves were found in the foamed quadrats in Winter 1. However, more brown leaves were present in the foamed quadrats in Autumn 2. In Summer, there were less yellow leaves in the canopy in foamed quadrats. These results are contradictory, but indicate overall that foaming does not adversely effect canopy health.

Table 4.12. Significant results for chi-squared tests for *Leucopogon collinus* canopy health

		CANOPY			YELLOWNESS			REDNESS			BROWNNESS		
season	n	chi	DF	P	chi	DF	P	chi	DF	P	chi	DF	P
UNBURNT													
autumn 1	87	6.50	2	0.039 F < C									
winter 1	85										4.60	1	0.032 F < C
summer	74				5.64	1	0.018 F < C						
autumn 2	66										4.24	1	0.039 F > C

Xanthorrhoea australis

X. australis resprout width in foamed quadrats decreased significantly from Winter 1 to Spring, whereas width stayed the same size in unfoamed quadrats (Table 4.13). There were no other differences in canopy height, width or area in any season in either the burnt or unburnt treatment (Figure 4.9).

The effect of foaming on canopy health was to have a smaller but more healthy canopy (Table 4.14). In Spring in the unburnt treatment, and Winter 2 in the burnt treatment, there were less yellow leaves in the foamed quadrats, i.e. a more green and healthy canopy. The canopy fullness was found to be lower in burnt and foamed quadrats in Summer.

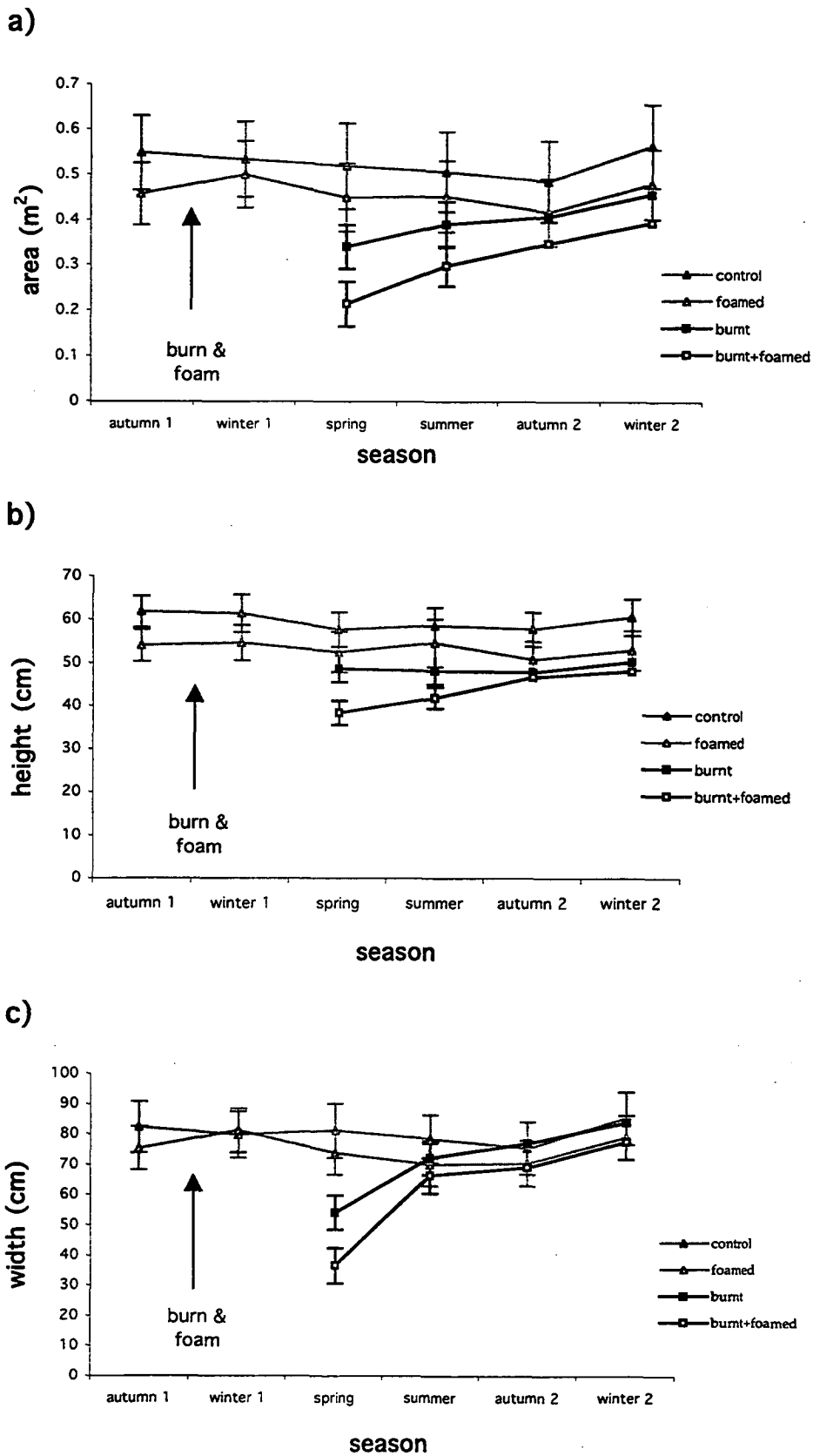


Figure 4.9. *Xanthorrhoea australis* canopy a) height, b) width and c) area over time for burnt and unburnt treatments. Standard error bars are shown. Arrow shows timing of burn and foam treatments

Table 4.13. Results for one-way ANOVAs for the change in *Xanthorrhoea australis* canopy height, width and area from season to season. Figures show mean values for height and width in cm and for area in m^2, followed by standard error in brackets

AREA	F	C	P		BF	B	P	
aut1-wint1	0.020 (0.019)	-0.015 (0.025)	0.212	ns				
wint1-spr2	-0.048 (0.019)	-0.014 (0.020)	0.202	ns				
spr2-sum2	0.002 (0.035)	-0.013 (0.019)	0.932[1]	ns	0.068 (0.047)	0.056 (0.041)	0.786	ns
sum2-aut2	-0.034 (0.035)	-0.020 (0.017)	0.671[1]	ns	0.045 (0.019)	0.017 (0.021)	0.306	ns
aut2-wint2	0.060 (0.011)	0.078 (0.015)	0.508	ns	0.026 (0.017)	0.049 (0.026)	0.400[1]	ns
HEIGHT	**F**	**C**	**P**		**BF**	**B**	**P**	
aut1-wint1	-1.739 (2.465)	-0.412 (3.079)	0.130	ns				
wint1-spr2	-2.043 (1.356)	-3.647 (1.680)	0.158	ns				
spr2-sum2	2.043 (2.834)	0.882 (1.770)	0.352	ns	4.087 (5.056)	1.033 (3.627)	0.509	ns
sum2-aut2	-3.696 (3.053)	-0.765 (1.381)	0.466	ns	4.348 (1.282)	-0.200 (1.919)	0.094	ns
aut2-wint2	2.130 (0.997)	2.765 (1.113)	0.762	ns	-0.826 (1.926)	2.400 (1.796)	0.434[1]	ns
WIDTH	**F**	**C**	**P**		**BF**	**B**	**P**	
aut1-wint1	2.391 (2.987)	-2.471 (1.925)	0.065	ns				
wint1-spr2	-7.087 (1.724)	1.294 (2.703)	0.008	**				
spr2-sum2	-3.609 (3.100)	-2.647 (1.981)	0.469[1]	ns	28.435 (8.186)	19.900 (5.577)	0.076	ns
sum2-aut2	0.435 (2.028)	-3.059 (2.486)	0.288	ns	2.522 (2.804)	4.933 (2.685)	0.712	ns
aut2-wint2	8.174 (1.520)	9.882 (2.427)	0.758	ns	4.348 (2.379)	6.767 (3.049)	0.404	ns

[1] result from Kruskal-Wallis rank order test. ns, not significant; *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$

Table 4.14. Significant results for chi-squared tests for *Xanthorrhoea australis* canopy health

		CANOPY			YELLOWNESS			REDNESS			BROWNNESS		
season	n	chi	DF	P	chi	DF	P	chi	DF	P	chi	DF	P
UNBURNT													
spring	39				4.93	1	0.026						
							F < C						
BURNT													
summer	51	6.10	2	0.047									
				F < C									
winter 2					3.86	1	0.049						
							BF<B						

Animal scats

No animal scats were found in the unburnt treatment. Rabbit and wallaby scats were significantly higher in the burnt compared with the unburnt treatment for all seasons scats were counted (Summer, Autumn 2 and Winter 2). Wombat scats were only significantly higher for the Summer score (the first count after the initial removal of scats in Spring).

Foaming did not have any effect on the total number of scats or on the scats of the individual animals (Table 4.15). There were no significant differences in scat numbers between foamed and unfoamed burnt quadrats.

Table 4.15. Results for one-way ANOVAs for differences in numbers of animal scats in burnt treatment between foamed and unfoamed

BURNT	BF	B	P	signif
RABBIT				
summer	5.47 (2.16)	14.33 (5.31)	0.817[1]	ns
autumn 2	7.60 (2.99)	9.87 (4.13)	0.699	ns
winter 2	26.73 (8.33)	14.07 (3.25)	0.296	ns
WALLABY				
summer	40.73 (5.29)	40.47 (3.47)	0.243	ns
autumn 2	26.53 (3.19)	27.07 (3.13)	0.920	ns
winter 2	38.40 (4.70)	40.20 (5.21)	0.461	ns
WOMBAT				
summer	0.73 (0.41)	0.40 (0.24)	0.471[1]	ns
autumn 2	0 (0)	0.20 (0.14)	0.150[1]	ns
winter 2	0.67 (0.47)	0.33 (0.27)	0.609[1]	ns
ALL				
summer	46.93 (5.71)	55.20 (6.74)	0.470	ns
autumn 2	34.13 (4.02)	37.13 (5.02)	0.888	ns
winter 2	65.80 (9.24)	54.60 (6.26)	0.510	ns

[1] results from Kruskal-Wallis test. ns, not significant

DISCUSSION

Effects of fire-fighting foam and burning

The main effect of foaming in the burnt treatment was a relative depression in *Banksia marginata* canopy growth over the main growing period from Spring to Summer. This is similar to the growth retardation of *Tsuga heterophylla* in the first year after foams were applied in an experiment by MacMillan Bloedal (Norecol 1989). Another adverse influence of foams seen by other researchers was a decrease in shoot growth of *Symphoricarpos occidentalis* (Larson and Newton 1996). It is possibly the surfactant properties in the foam that causes such undesirable effects (Adams 2000). In contrast, no detrimental effects on the growth characteristics of *Banksia integrifolia* (Coast Banksia) occurred after the application of varying concentrations of a Class A foam (Hartskeerl 1999). However, in this glasshouse experiment, fire was not included.

Fire retardants can inhibit the growth of North American species, but it is inappropriate to extrapolate their effects to those of the foam used in the present study for four reasons. Firstly, retardants generally have a higher level of the fertilising salts ammonium phosphate and ammonium sulphate, whereas the ForExpan-S used in the current study had levels of phosphorus and nitrogen well below fertiliser concentrations — even in its concentrated form — and was then diluted to 1% with water. Secondly, the North American species shown to be affected by retardants are quite different to fire-prone

sclerophyllous Australian plant species. Thirdly, Californian annual grasslands and North Dakota prairies would have a higher natural fertility than the low-nutrient coastal heath in the current study. And, finally, the effects of fire that were included in the present study were not always included in the North American studies. The one Australian study of the effects of fire retardants on a mixed eucalypt and *Angophora costata* forest showed that adult plants in the family Proteaceae generally did not decline in cover (Bradstock *et al.* 1987).

Other, more minor, vegetation responses in the burnt treatment were short-term reductions in resprout canopy fullness in *Banksia marginata* towards the end of the experimental period and in *Xanthorrhoea australis* in the summer after the burn. Both species resprouted vegetatively after the fire. Bradstock *et al.* (1987) found similar short-term reductions in canopy cover of *Banksia spinulosa* (Hairpin Banksia) and their *Xanthorrhoea* 'species 1' after treatment with a retardant but no burn. Given the short-term nature of the response and the ability of both *Banksia* species and *Xanthorrhoea* species to survive the defoliation of fires (Gill and Ingwersen 1976), the small reduction in canopy fullness should not cause long-term problems to the heathland ecology.

Another result was an increase in canopy size of *Aotus ericoides* in foamed and burnt compared with burnt only quadrats, during the whole experimental period. It is unclear how much this result was an affect of the treatment, because canopy size was generally larger (although not significantly so) prior to the application of the foam treatment. However, there was also an increase in the rate of growth from one season to the next in burnt and foamed quadrats. The rate of growth does not necessarily depend on the size of the canopy, as canopies may be larger because they are older rather than because they grow faster. Only one other researcher has found that foaming increases growth; Larson and Newton (1996) measured increased leaf length in *Symphoricarpos occidentalis*.

The short-term reduction in species richness measured in the current study is in accordance with the decreases reported by Larson and Newton (1996). The fact that the two types of foam used were different (ForExpan-S in the current study and Silv-Ex in the study by Larson and Newton) need not hinder a direct comparison as the chemical compositions appear quite similar (Norecol 1989; Angus Fire Armour 1997; Hamilton *et al.* 1998; Adams and Simmons 1999). When lifeform groups in the current study were tested individually for differences in species richness, the only significant decrease was found in ferns — a result that could be due to the differences which were already present in the baseline data. However, graminoids, herbs and orchids all showed lower richness in foamed compared with unfoamed quadrats and, although none was individually significant, the addition of all together probably explain the overall species richness decline. This decline was short-lived, lasting only for the Spring after the burn.

One interesting finding in the current study is that there was no increase in the numbers of exotic grasses or herbs in the period of the experiment. The small number of exotics found in the experimental area, or within dispersal distance, and the small amount of nutrients added in the foam probably explain this result. There was, however, a weakly significant increase in native grass species in autumn, which is similar to the increase in growth of the weedy grass *P. pratensis* found by Larson and Newton (1996). Both results show a potential 'fertiliser effect' of either phosphorus or nitrogen. *P. pratensis* has also been suggested to respond more positively to the disturbance caused by burning in a mesic environment (Larson and Newton 1996).

There were no differences in leaf colour between foamed and unfoamed quadrats in the burnt treatment, unlike differences found in the unburnt treatment, as discussed below. One reason is likely to be the lack of any canopy when the foam treatment was applied after all the vegetation had been burnt. Another reason may be that vigorous growth after burning swamps any canopy ill effects caused by foaming. Finger *et al.* (1997) noted that burning obscured any responses by vegetation to fire-fighting chemical applications that were seen in unburnt sites.

Unlike the preferential grazing by cattle of retardant-treated areas compared with control and burnt-only areas (Larson and Duncan 1982), the current study did not find any difference in native or feral animal presence (as measured by the number of scats) in foam-treated quadrats. The attraction of animals to the green pick found after burning is likely to be a far more influential factor.

Insect damage was not measured in the current study, although it was noticed during fieldwork that *B. marginata* leaves showed evidence of insect attack, in some cases quite severe. It would have been an interesting effect to quantify, as other researches have found increased herbivory associated with Silv-Ex-treated vegetation (Larson and Newton 1996), elevated levels of foliar nitrogen (Landsberg *et al.* 1990) and fertilising with phosphorus (Specht 1963). Adams and Simmons (1999) point out that increased insect attack on retardant or foam treated plants may have negative effects on the regrowth of vegetation after fires.

Effects of fire-fighting foam without burning

As with the burnt treatment, *Aotus ericoides* responded with increased canopy size when foamed in the unburnt treatment. The effect remained, to a greater or lesser degree, for the entire period of the experiment. As with the burnt treatment, this result is not necessarily attributable to the foam because of the initial difference between treatment groups before the foam was applied. However, there were also significant increases in growth from one season to the next in foamed quadrats which could conceivably be a treatment effect.

As discussed above, increased growth of native Australian species is not an effect usually found from either foams or retardants. Hartskeerl (1999) found no effects of foams on various growth characteristics of eight species, including two in the Fabaceae family. And all five species of Fabaceae showed declines in cover in the mixed eucalypt – *Angophora costata* forest in New South Wales (Bradstock *et al.* 1987). Increased growth could be a response to increased nutrients. Although the actual amount of phosphorus added in the foam was very small, the ratio of phosphorus to nitrogen could explain the growth increases by *A. ericoides* in the current study. Heathland vegetation is generally thought to be phosphorus-poor (Specht 1963; Specht and Groves 1966), although there is some evidence that the growth of fynbos shrub species is limited by nitrogen (Witkowski *et al.* 1990b). In the current study, the ratio of phosphorus available in the soil (added in the fire-fighting foam) to the amount of nitrogen in the soil (from the addition of fire-fighting foam and the nitrogen-fixing properties of *A. ericoides*) may have been in the correct proportions to allow increased growth of *A. ericoides* but not non-nitrogen fixing species. The addition of water as well as foam may have caused a positive growth response. However, given the free-draining nature of the sandy soils, such increased growth would not be expected to last very long.

The other four species showed a decrease in canopy size in foamed quadrats, a result more in line with other authors' results. *Banksia marginata* canopies had a relative depression of growth from Spring to Summer, whereas there was an absolute decrease in width in *Xanthorrhoea australis* canopies from Winter 1 to Spring. In *Leptospermum scoparium* the canopy width decrease persisted to the end of the experiment but the *Leucopogon collinus* canopy width decrease appeared only in the last two seasons of the experiment. Bradstock *et al.* (1987) measured declines in cover scores not just for the *Banksia* and *Xanthorrhoea* species mentioned above, but also for two *Leptospermum* species and for *Epacris pulchella* (NSW Coral Heath), an Epacridaceae, due to the application of a retardant. This comparison must be made with caution because of the differences between retardants and foams, and also because there is no obvious trend in susceptibility to retardants based on taxonomic or lifeform grouping (Bradstock *et al.* 1987). Again, it is possibly the surfactant properties in the foam that would cause declines in growth (Adams 2000).

There was no difference in total species richness between foamed and control plots, however, grass species in foamed quadrats decreased in Summer, the opposite result to that found in the burnt experiment. Whatever competitive advantage was present for grasses in foamed and burnt quadrats does not appear to have been present in foamed only quadrats. As with the burnt experiment, foaming did not appear to have any effect on exotic species richness.

When foam was added to quadrats in the burnt experiment, there was no plant canopy present, therefore any response from plants is most likely due to a change in soil properties

caused by the addition of foam. In the unburnt experiment, two responses are possible: the direct effect of foam contact to the canopy and the more indirect effect of soil nutrient changes. Effects on the colour and health of the plant canopy were seen in all five species measured, although the most significant response was seen in *B. marginata*.

Foaming caused chlorosis, reddening and dead leaves in the canopy of *B. marginata*, occurring for the entire period of the experiment. This is in common with other studies showing leaf edge browning (Norecol 1989), and senescence and shoot damage (Larson and Newton 1996) after application of Silv-Ex, and short-term blight and yellowing in trees due to foam applications (Norecol 1989). In Australia, glasshouse experiments using a retardant solution painted directly onto leaves resulted in damage and death to branch tips and young leaves of *Banksia collina* (Hairpin or Hill Banksia) seedlings (Bradstock *et al.* 1987). Six weeks later, all plants had virtually fully recovered through the production of new leaves. Ammonium sulphate was considered the ingredient responsible for the damage, although the mechanism was not known (Bradstock *et al.* 1987). Other research has shown no effect of a class A foam on the leaf colour of *B. integrifolia* (Hartskeerl 1999).

The mechanism for leaf damage to *B. marginata* in the current study could be a direct alteration of the chemistry of the leaf. Bradstock *et al.* (1987) suggest that the pattern of foliar damage found in their glasshouse experiments was similar to that for salt-spray affected plants. Moodie *et al.* (1986) showed a relationship between surfactants entrained in airborne seaspray and increased uptake of salt in the foliage of *Araucaria hetrophylla*. Deterioration consisted of browning and loss of green foliage, gradually extending over the entire crown and epicormic shoots. Gellini *et al.* (1983) also found an increased level of damage from the combination of surfactants and salt on the needles of *Pinus pinea* (Italian Stone Pine). In particular, chlorosis was concentrated at the needle tip and necrosis occurred at high concentrations. Given the presence of surfactants in ForExpan-S and the proximity to the coast this is one possible explanation for the *B. marginata* leaf damage found in the current study.

Another possible explanation is an imbalance of trace elements or other nutrients in the soil, resulting from the application of the fire-fighting foam. Addition of fertilisers in heathland resulted in marked leaf yellowing in *Banksia ornata* (Desert Banksia) (Specht 1963), and yellowing in *Leptospermum myrsinoides* (Heath Teatree) leaves (Heddle and Specht 1975). However these results were for large quantities of fertiliser and were over a long time period (the original experiment was revisited some 22 years later).

The results for the other four species show either greener and more healthy foliage in foamed quadrats or inconclusive results. *L. scoparium* and *X. australis* both showed less chlorosis in foamed quadrats in the spring after the treatment was applied. *A. ericoides* showed less chlorosis directly after the foaming treatment but more chlorosis in foamed

quadrats two seasons later. *L. collinus* also showed less chlorosis initially (in summer), then more canopy browning in the following autumn. None of these responses occurred directly after application of the foam but with a delay of at least one season. The different pattern of responses suggest another mechanism than that causing *B. marginata* foliage damage. One possibility is that post-treatment weather conditions influenced the foam effects. Larson and Newton (1996) and Hamilton *et al.* (1998) explain the different responses by the same plant species in North Dakota prairies compared with Great Basin shrub steppes as due to departures from the usual rainfall patterns. The tree death after application of a 1% solution of Silv-Ex seen by the Canadian Forestry Service (described by Norecol (1989)) did not occur when the coating was washed off two days later. Bradstock *et al.* (1987) found that *Acacia longifolia* (Sallow Wattle) seedlings were responsive to washing whereas *Banksia collina* seedlings were not. The ability of any species to survive after an application of retardant or foam may be a function of the presence and vigour of regenerating buds and other species growth characteristics (Bradstock *et al.* 1987).

The only effect on flowering seen in the current study was a reduction in the overall number of flowers in autumn for *Leptospermum scoparium*. It is not know if this was a temporary phenomena or one that would continue into the next flowering period. This result is similar to that for *Sprengelia incarnata* after application of Silv-Ex found by Marsden-Smedley (1992). In contrast, large amounts of fertilisers added to heathland in South Australia resulted in increased numbers of flowers and an earlier timing of flowering in *Leptospermum myrsinoides* (Specht 1963).

Breakdown of foams

The fire-fighting foam ForExpan-S is made up of hydrocarbon surfactants, glycol solvents and water. Each of these components breaks down slightly differently. Hydrocarbon surfactants break down by oxidisation or other alteration by bacterial action, but the main degradation pathway is essentially biological. The adsorption of the surfactant to soils is dependent on the silt, clay and organic matter content and the surface area. In very sandy soils such as found under heathland at the Mt William National Park, the mostly impervious minerals in the sand would generally be weak adsorbents (Norecol 1989). The solvent hexylene glycol mineralises using a CO_2 evolution procedure, taking about 16 days to degrade by 50% and 42 days to degrade to over 85% (Norecol 1989). Many retardants and foams (e.g. Phos-Chek, Fire-Trol and Silv-Ex) degrade more quickly in soils with high organic content than in soils of low organic content (such as found at the study site). These differences in breakdown rate indicate that in the acid, low organic matter sandy soils found in this experiment breakdown may be quite slow. Therefore any foams applied can

possibly enter the groundwater through runoff, if sufficient rainfall occurred before breakdown was complete, thus extending possible environmental impacts.

CONCLUSION

The effects of the Class A fire-fighting foam ForExpan-S on the coastal heathland in the Mt William National Park range from positive to negative and short term to longer term. A shrub in the Fabaceae family, *Aotus ericoides*, showed increased growth when foamed and when burnt and foamed, a result not found by other researchers. Minor decreases in canopy size in *Banksia marginata* and *Xanthorrhoea australis* when burnt and foamed, and *Leptospermum scoparium* and *Leucopogon collinus* when foamed, are results more in line with biomass decreases shown to occur in other species and communities after foaming. Leaf damage is a response found by other researchers and the present study indicates *B. marginata* leaves are particularly susceptible to foaming. A short-term decline in total species richness and increase in grass species richness was found after foaming and burning. Flowering can be reduced in the short-term, a result found for *L. scoparium* in the present study, in line with the result for *Sprengelia incarnata* found for buttongrass moorland in Tasmania. The present study establishes that fire-fighting foam does have some impact on heathland plants, although the longer term impact on population dynamics was not able to be tested.

However, given these results, and the slow breakdown process of foams in sandy soils, the use of foams in heathlands in Tasmania needs to be carefully considered. Fire-fighting foams are obviously a valuable tool, but their effectiveness at suppressing fires, the safety of the fire-fighters and the cost of fire-fighting equipment all have to be balanced with their potential effects on the heathland ecosystem, the value of that heathland and the environmental effects of other suppression techniques.

5. Factors influencing the condition of coastal heathland remnants in Tasmania

INTRODUCTION

European colonisation of Tasmania brought about great changes in a landscape previously managed by Aboriginal people. Heathlands were relatively undisturbed by Europeans for over 100 years as areas of higher fertility were cleared and converted to agriculture. Following the discovery in the late 1930s of the beneficial effects of trace elements on pastures and cereals, a method was provided by which large areas of heathland could be cleared, ploughed, fertilised and successfully sown to pasture or grain (Kirkpatrick and Harris 1999b). By 1995, approximately 200,000 ha of coastal heathland, or nearly half the pre-European amount, had been cleared in Tasmania (Kirkpatrick 1991). Clearing continues to be one of the main threats to the remaining areas of coastal heathland.

Where areas of heathland meet areas of cleared vegetation, there is an edge or boundary that acts as an interface between the native vegetation and the exotic vegetation. Boundaries usually have quite different micro-climates compared with the interior, due to changes in wind action, water movement, radiation balance and evapotranspiration (Ranney *et al.* 1981). Boundaries can also act as a filter for disturbances and exotic species, or provide a source for biological invasion of the interior native vegetation (Sisk and Margules 1993). In heathlands, boundaries will be subject to many disturbance, for example, increases in nutrients, grazing and trampling from herbivores, fungal pathogens, fire and, most importantly, invasion by exotic species.

Research into the effects of nutrient accession to heath edges has been undertaken on fertiliser drift and crop residues (Muir 1979; Grigg *et al.* 2000), oxides of nitrogen from vehicle exhaust (Angold 1997), imported fill material (Clemens and Franklin 1980), and urban runoff (Clements 1983). In all cases, the input of nutrients raised either phosphorus or nitrogen levels by measurable amounts in the heathland and in some cases an increased growth in heath species was observed. The most significant result of the raised nutrient levels was an increase in grasses, exotic species and/or mesophytic species at the expense of scleromorphic shrubs. Where heathland is adjacent to regularly fertilised farmland, there is a real risk of nutrient accession and a resultant decrease in the integrity and condition of the heath.

Alpine heaths and heathy woodlands have been the focus for research into the effects of grazing on heath edges. It is hard to summarise these effects due to the different vegetation types studied, the different herbivores studied (e.g. native mammals, feral mammals, domestic stock or invertebrates) and the interactions between grazing and other

types of disturbance (e.g. fire, clearfelling). In general, the abundance of some plant species will be reduced relative to others, due the different feeding preferences of the different herbivores (Kirkpatrick and Harris 1999a). Mammal browsers will usually prefer the grasses and herbs found in pasture over the scleromorphic shrubs and silica-rich graminoids found in the heath. However, the heathland does provide some palatable leguminous shrubs, and also provides shelter and cover for both native herbivores and domestic stock. The most common effects of herbivores seem to be a decrease in shrub cover, density and/or diversity (Leigh and Holgate 1979; Dickinson and Kirkpatrick 1986; Williams 1992; Scougall *et al.* 1993; Pettit *et al.* 1995), an increase in bare ground (Leigh *et al.* 1987), and an increase in exotic species (Scougall *et al.* 1993; Pettit *et al.* 1995). To date there has not been any research focussing on the effects of grazing on coastal heath in Tasmania.

The root-rotting fungus *Phytophthora cinnamomi* is widely established and continually expanding in Tasmania (Podger *et al.* 1990b). Tasmanian heathlands occur in the broad environmental conditions preferred by the pathogen — an average rainfall of at least 600 mm and a mean annual temperature of greater than 7.5° C (Podger *et al.* 1990b). Some heathlands also experience the alternating periods of waterlogging and water stress which create the ideal disease conditions for the fungus (Malajczuk and Glenn 1981). Many of the species commonly found in coastal heathlands, including members of the families Epacridaceae and Proteaceae, are very susceptible to *P. cinnamomi*. A large proportion of the susceptible species are rare or threatened in Tasmania (Barker and Wardlaw 1995). The pathogen has caused significant modification of and severe damage to heathlands (Podger *et al.* 1990b), in some cases leading to dieback of heathy understoreys and an influx of exotic species and native grasses (Weste 1974). Where there is a ready source of exotics, such as farmland, adjacent to heaths suffering from *P. cinnamomi* infection, weed invasion has the potential to increase.

Fire management can have a dramatic effect on the integrity and condition of heathland in Tasmania. Too frequent burning can lead to a decline in presence or cover of obligate seeders in heathlands on the Australian mainland (Lewis and Harshbarger 1976; Nieuwenhuis 1987; Cowling *et al.* 1990; Morrison *et al.* 1996; Bradstock *et al.* 1997). Although in Tasmania there are fewer species that respond in this way, there are still some that have been reduced or eliminated by frequent fires (Kirkpatrick and Harris 1999b). On the other hand, too long an interval between fires can result in a transition from a highly diverse heathland to a species-poor scrub (Specht *et al.* 1958; Molnar *et al.* 1989; Morrison *et al.* 1995). The interaction between fire frequency, nutrient addition and weed invasion in heathlands is at the moment poorly understood.

All these disturbance factors: nutrient increase, grazing, fungal pathogens and fire, will have an influence on the integrity and condition of heathlands, as measured by the

invasion of exotic species. Other factors, such as salt spray and drought, will also affect the condition of heath but are not so directly under human control. Invasion by exotic plants into a variety of native communities is a problem for conservation in Australia and has received much attention, particularly the importance of disturbance (Amor and Piggin 1977; Fox and Fox 1986; Hobbs and Atkins 1988; MacDonald *et al.* 1988; Hobbs 1989). In heathlands, exotic species have become established following the addition of fertiliser (Connor and Wilson 1967), due to changes in fire regime (Hopper and Muir 1984; MacDonald *et al.* 1988) and as part of the rehabilitation process after sand mining (Thatcher and Westman 1975). Current ecological research into the effects of habitat fragmentation in heathland needs now to be directed at managing and controlling the external influences that lead to such decreases in integrity and condition.

This chapter investigates which environmental, management and disturbance factors influence the condition of coastal heathland remnants in Tasmania. In an effort to understand the main factors influencing the penetration of exotic species into heathland, this survey aimed to sample as many different variables as possible. The heath–pasture boundaries were chosen to cover a wide range of features such as age, environmental variables, type of cleared area, management regimes, geographical location and wet or dry heath. Toward this aim, a larger number of sites were visited once, rather than fewer sites more often.

STUDY SITES

The study was restricted to lowland coastal heaths of the north and east coasts of Tasmania. The buttongrass moorlands of the southwest, heathland in the far northwest of the State and alpine heathlands were not included.

Potential boundaries were first identified from comparison of cleared areas on satellite images with the heathland maps of Kirkpatrick (1977). For Flinders Island, possible boundaries were mapped at a smaller scale using aerial photographs. Opinions of land managers, National Parks rangers and other people with detailed knowledge of local vegetation conditions were sought to verify the presence of a boundary or to suggest other locations.

Once as much information was gained as possible, a field trip was undertaken to find the boundary and decide its suitability for the survey. The essential characteristics of a boundary was that it was a true heathland or woodland with heathy understorey (scrub over 2 m tall was not included) adjacent to an area cleared by human activity. Complicating factors such as changes in soil type, slope or drainage between heathland and cleared area were avoided. Large dirt tracks or sealed roads at the boundary were also avoided as it was felt they would interfere with the target edge effects. However, some

small, low-traffic vehicular tracks and foot or animal tracks at the boundary were unavoidable. A desirable characteristic was a simple or sharp boundary that was easily defined to aid in the calculation of depth of penetration by weeds. In reality this requirement was difficult to fulfil because few boundaries were simple, so all types of boundaries were accepted and the boundary features were included as a variable.

Fifty transects were surveyed in 33 different sites, over the period August 1997 – August 1998 (Figure 5.1). The coastal heathland surveyed was most commonly on areas of very low slope and ranged in altitude from 5 to 120 m. The surrounding land use was predominantly sheep or cattle grazing or shack settlement. Approximately half the sites had the heathland section of the transect in a reserve, reflecting the fact that some 15% of the original area of lowland heath was reserved by 1995 (Kirkpatrick and Harris 1999b).

METHODS

Sampling design and measurements

Transects

Transects were 20 m long and situated perpendicular to the heath–pasture boundary. Six 1 m x 10 m quadrats were placed along the transect with the long edge parallel to the boundary (Figure 5.2). A long, skinny quadrat was used to provide a long edge for the line intercept measure (described below) and to reduce the damage from trampling. The size used (10 m^2) was considered the smallest size that would adequately pick up most species present, yet the largest size that would not become too cumbersome in the (often) dense heathland. One quadrat was put in the pasture (P), one at the very edge of the heath (A) and the remaining four continuing into the heath with the distance between them doubling (B–E) (Figure 5.2). This allowed for coverage further into the heath than if the distance between quadrats had been the same amount (e.g. 1 m). The deepest quadrat into the heath was thus at 20 m. Past 20 m the heath was scouted for exotics along a hypothetical continuation of the transect line but most exotics further than 20 m tended to be found along native mammal trails, therefore it was felt that 20 m was adequate for this survey. A separate native mammal trail weed survey was initiated to test whether there were significantly more exotics along the trails. Where the boundary was not distinct (more than one fenceline, or a combination of fence, track and ditch), extra quadrats were placed at the edge of the heath or pasture.

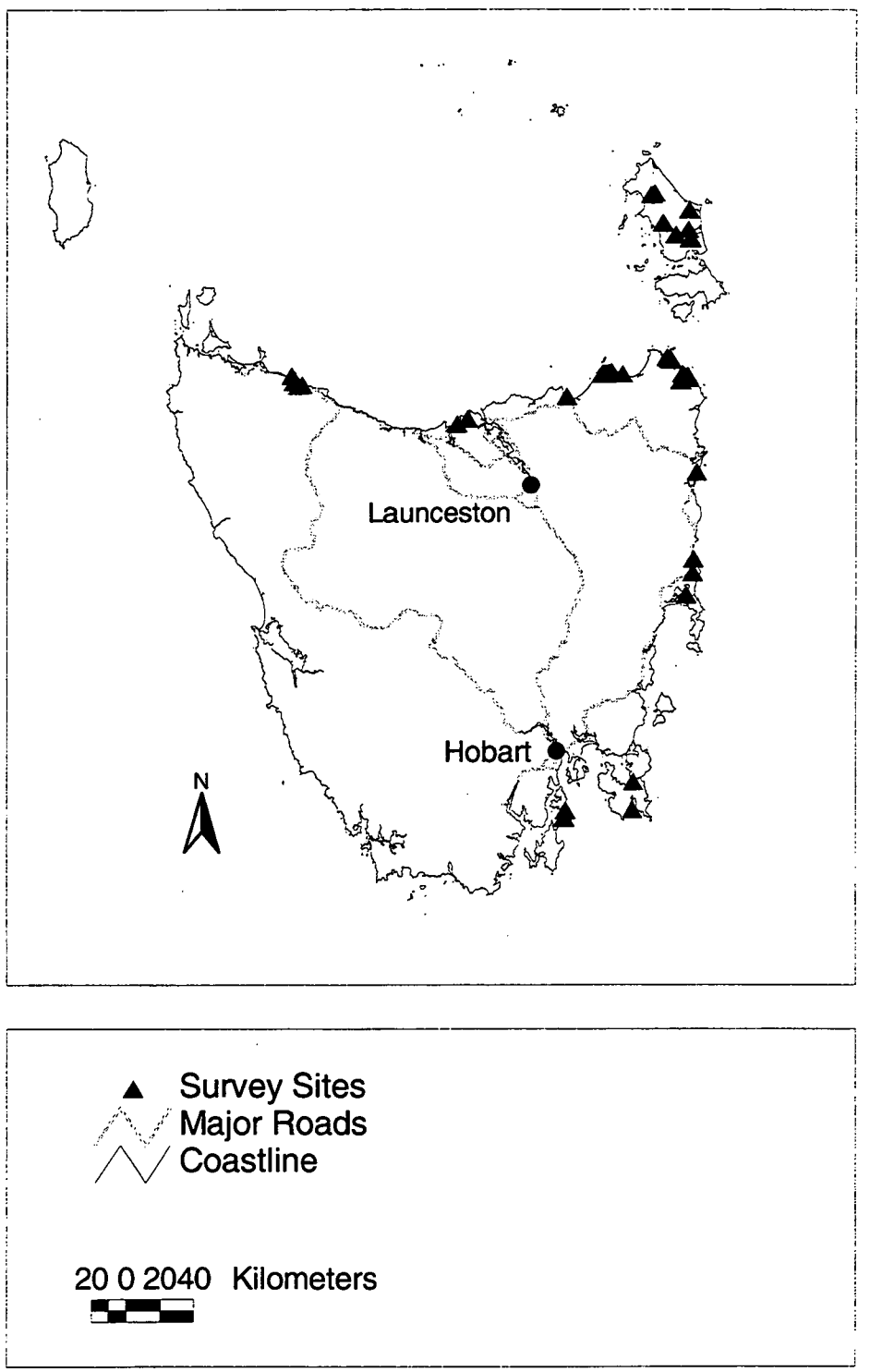


Figure 5.1. Location of heath–pasture boundary sites in Tasmania

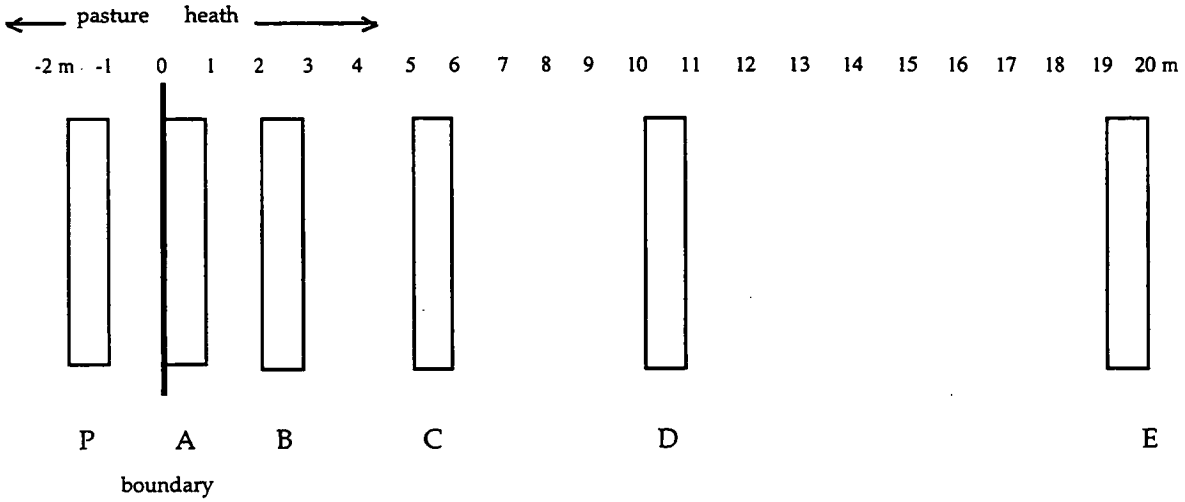


Figure 5.2. Schematic representation of quadrat placing respective to heath edge

Floristic and structural variables

All vascular species in the quadrat were identified and given a cover value using a modified Braun–Blanquet scale (1, <1%; 2, 1–5%; 3, 6–25%; 4, 26–50%; 5, 51–75%; 6, > 75%). Species that were not able to be identified during the fieldwork period were either excluded or grouped into larger categories. For example, many grasses could not be identified because there was inadequate reproductive material, so were identified to genus level only. In some cases a plant could only be identified by its plant lifeform category and whether it was exotic or native. Due to the wide range in distance and time of the survey, the lowest common denominator dictated to what degree a species could be identified.

Plant cover in lifeform categories and the amount of scats and bare ground were assessed using the line-intercept method described in Mueller-Dombois and Ellenberg (1974). The 10 m edge of the quadrat closest to the fenceline (heath quadrats) or furthest from the fenceline (pasture quadrat) was used for the line intercept. The lifeform categories were *Xanthorrhoea australis* (Austral Grasstree), fern, native grass, native graminoid (monocotyledonous plant other than grass), native herb (non-woody dicotyledonous plant), native shrub (woody dicotyledonous plant), native tree, exotic grass, exotic graminoid, exotic herb, exotic shrub and exotic tree. Projective cover for each category was measured, often resulting in a total cover of over 100%, reflecting the overlapping vegetation structural layers. Species nomenclature follows that of Buchanan (1999).

The height of each major vegetation structural class was measured, and the structures were divided into three layers: low, 0–49 cm; medium, 50–100 cm; and high, 101+ cm. The dominant and subdominant species was recorded for each class — if there was more of one species than any other in that class, and its cover was more than 5%, it was recorded as the dominant species. If there was another species that had more than 50% cover of the

dominant species it was recorded as the sub-dominant species. The basal area of tree species was measured using the Bitterlich Variable Radius Method (Mueller-Dombois and Ellenberg 1974).

Environmental and disturbance variables

For each transect, the grid reference, altitude and land tenure were collected from topocadastral maps. The variable 'region' was created by grouping the transects by geographical region so that there were at least 5 transects in each group (Table 5.1). During fieldwork, the type of heathland boundary (e.g. fence, ditch, track), and the type of cleared area (e.g. pasture, garden, oval) were noted. A boundary index was developed by adding the number of different boundary features (i.e. 2 fences and 1 track would give a boundary index of 3, no fences, tracks or ditches would give an index of 0). The age of the boundary was determined from aerial photos and/or interviews with landowners.

Some site data were collected for three of the six quadrats only, which were the pasture quadrat (P), the heath edge quadrat (A), and the innermost heath quadrat (E). These data were geology, slope, aspect, alignment of boundary (i.e. along which compass direction the long edge of the quadrat was aligned, sometimes different to the aspect), position in landscape (top of ridge, upper slope, lower slope, flat) and soil variables. For each soil sample the colour was assessed using the Munsell Soil Colour Chart and three aspects of the colour were used as variables: hue (the dominant spectral colour e.g. red or yellow), value (the relative lightness) and chroma (degree of saturation) (Munsell Soil Color Company 1954). The texture was assessed in the field using the technique of McDonald *et al.* (1984). A CSIRO Inoculo Laboratories field test kit was used to measure pH in the A horizon. A 45-mm diameter metal soil corer was used to take soil samples from five points along each of the three quadrats and the samples were then bulked in impermeable plastic bags. The bagged samples were opened and left to air dry for 2 weeks, then ground and sieved to 500 μm. Phosphorus was analysed by Bray's extractable phosphate method using an acid ammonium fluoride extractant (method 2) and total nitrogen using Kjedahl digestion (Jackson 1958).

A series of disturbance variables were collected for each quadrat. Disturbances noted were presence or absence of animal tracks, human foot tracks or vehicle tracks, native or exotic animal grazing, native animal diggings, stock trampling, fallen branches or logs, cut or damaged plants, cut stumps or logs (indicating firewood collection), slashing, clearing, wind pruning, soil disturbance (piles of soil, erosion, compaction, ripping, waterlogging) and dumping of rubbish. A disturbance index was calculated by scoring 1 for each of the above categories, then adding all these scores together. For native mammal grazing, the amount of grazing (1, low; 2, medium; 3, heavy or severe) was scored for each of wallabies,

Table 5.1. Number of sites occurring for each of the categorical variables

characteristic	number of transects
Environmental variables	
region	
East Coast	5
Flinders Island	10
Northeast	11
Northwest	6
Southeast	7
Waterhouse	11
aspect	
northwest, north, west	10
southwest, northeast	22
south, east	13
southeast	5
alignment	
north-south, northwest-southeast	24
west-east	20
southwest-northeast	6
geology	
quaternary sands	32
limestone	5
other (granite, dolerite, mudstone)	13
landform	
upper slope and ridgetop	10
lower slope	10
flat	30
slope class	
0-1 degrees	28
2 degrees	7
3-5 degrees	8
6 or more degrees	7
Soil variables	
edge soil hue class	
red-5YR	5
7.5YR	24
10YR-5Y	20
edge soil chroma	
chroma 1	39
chroma 2-4	14
edge soil texture	
sand and clayey sand	8
loamy sand	20
sandy loam	17
other (loam, sandy clay loam, clay loam)	8
centre soil hue class	
red-5YR	15
7.5YR	13
10YR-5Y	21
centre soil chroma	
chroma 1	39
chroma 2-4	10
centre soil texture	
sand and clayey sand	6
loamy sand	23
sandy loam	15
other (loam, sandy clay loam, clay loam)	6
Pasture management variables	
age	
1-10 years	7
11-20 years	10
more than 20 years	27
land tenure	
freehold/private/leasehold	39
crown land/national park	11

characteristic	number of transects
Pasture management variables (cont'd)	
pasture type	
pasture	33
abandoned pasture	10
other (oval, garden, slashed)	7
pasture slashing	
not slashed	41
regularly slashed	8
Grazing variables	
years since grazed	
currently grazed	19
in last 1-5 years	15
more than 6 years ago	10
no. of years grazed	
1-5 years	21
6-10 years	8
11-20 years	5
more than 20 years	6
months in year grazed	
less than half the year	9
half the year - nearly all year	9
all year	22
season grazed	
all year	24
variable	17
cattle	
absent	24
present	24
sheep	
absent	19
present	29
other stock	
absent	42
present	6
stock class	
cattle	10
sheep	16
cattle and sheep	16
stocking rate	
1 or less dse/ha	6
2-5 dse/ha	9
6-20 dse/ha	10
> 20 dse/ha	9
wallaby	
light	12
moderate	11
heavy/severe	21
wombat	
none	10
light	12
moderate	8
heavy/severe	17
Cape Barren geese	
absent	39
present	8
feral grazing index	
none	15
light	16
moderate	16
Fertilising variables	
years since fertilised	
in last year	10
in last 2-5 years	15
in last 6-10 years	5
more than 10 years ago	9

Table 5.1. continued

characteristic	number of transects	characteristic	number of transects
Pasture management variables (cont'd)		**Heath management variables (cont'd)**	
years between fertilising		current grazing	
1 year or less	14	no grazing	17
2-10 years	10	rare	5
> 10 years OR only fertilised once	12	occasional/regular	12
season fertilised		previous grazing	
autumn only	28	not grazed	11
other season/regime	7	in last 5 years	15
fertiliser application		> 5 years ago	8
super spreader	24	type of stock	
other (direct drill, by hand)	8	no stock	11
lime		cattle	14
no	34	other	10
yes	6	season stock grazed	
Sowing variables		no stock	12
years since sown		all year round	11
1-5 years ago	8	parts of the year only	11
6-10 years ago	6	**Disturbance variables**	
11-20 years ago	13	fence	
more than 20 years ago	10	no fences	17
no. of times sown		1 fence	28
sown once only	29	2 fences	5
sown more than once	16	track	
season sown		absent	38
autumn	29	present	12
other (spring, summer)	5	ditch	
cocksfoot		absent	45
absent	12	present	5
present	19	boundary index	
fescue		no boundary marker	13
absent	22	1 boundary marker	24
present	9	2 boundary markers	7
subclovers		3 or 4 boundary markers	6
absent	15	disturbance edge	
present	16	none or 1 type	9
no. of different seed		2 types	20
2 types	8	3 types	19
3 types	7	4 or more types	10
4 types	6	disturbance 2-3 m from edge	
5 types	8	none or 1 type	8
seed application		2 types	20
spreader	20	3 types	15
other (direct drill, by hand)	10	4 or more types	7
Heath management variables		disturbance 5-6 m from edge	
land tenure		none or 1 type	7
private/leasehold	19	2 types	22
crown land	7	3 types	14
national park	10	4 or more types	7
other reserve	10	disturbance 10-11 m from edge	
last fire		none or 1 type	10
1-10 years	6	2 types	19
11-20 years	24	3 types	15
> 20 years	20	4 or more types	6
heath cleared		disturbance heath centre	
yes	14	1 type	7
no	21	2 types	22
heath slashed		3 or more types	21
yes	5	*Phytophthora cinnamomi* edge	
no	35	absent	40
where cleared		present	11
not cleared	19	*Phytophthora cinnamomi* centre	
along boundary	10	absent	24
entirely or small area	7	present	26

wombats and Cape Barren Geese and these scores were added together to create a native animal grazing index. The same was done for rabbits and hares to create a feral animal grazing index. The presence of *Phytophthora cinnamomi* in the heathland was scored visually by assessing health or dieback of susceptible species. Podger *et al.* (1990b)'s list of highly susceptible Tasmanian species was used.

Native mammal trails

Eighteen native mammal trails were followed from the heath–pasture boundary to a maximum of 20 m (measured along the trail, not as a straight line from the boundary). At 2-metre intervals, a 1-metre stick was placed horizontally perpendicular to the trail, and all vascular plant species touching the stick were identified. This 1-metre line intercept usually consisted of 20–40 cm of trail and 30–40 cm of heath to either side. The boundary between the trail and the heath was very distinct, the trail was an obvious depression in the soil and often included bare ground. The native mammal trail survey was done in the Waterhouse and Mt William regions.

Management variables

The owner or manager of both the heathland and the pasture or cleared land was interviewed, where possible. A series of questions was asked, covering current and previous land management and the age of the boundary (i.e. when the area was first cleared). Questions were asked about the grazing regime: when the pasture was last grazed, the total number of years grazed altogether, how many months in the year grazed, the type of stock, the stocking rate, and the level of grazing by native and feral animals. Information was gathered about the fertilising regime: when the pasture was last fertilised, how often it was fertilised, the season of fertilising, how the fertiliser was applied, the level of phosphorus, and whether lime was applied. The sowing history was also obtained: when the pasture was last sown, how often it had been sown since first cleared, the season it was sown, which seeds were sown and how they were applied.

For the heathland, the questions covered land tenure of the heath, the time and season of the last fire, whether the heath had ever been cleared and/or sown, if and so where and how long ago, whether the heath had ever been grazed and if so, when and for how long, and whether the grazing was continuing.

It was not possible to get answers to all questions at all sites, as sometimes the land owner or manager was not available to answer them, or did not know or remember the answers. Once as much information had been collected as was possible, the results were grouped into categories and the categories were combined to ensure there were always 5 or more sites in any category. Variables were discarded if all the answers were in the same category or if

information was available for less than 30 sites. Where the sites were divided into super regions (discussed below), categories were recombined to ensure there were still 5 or more sites per category. In some instances this meant there were not enough sites available for that variable to be tested.

Analysis

Condition variables

Nine dependent variables are used as surrogates for the 'condition' of the heathland communities, following the method of Kirkpatrick and Gilfedder (1995). For each quadrat, these are native species richness, native species cover, exotic species richness, exotic species cover, exotic species richness as a percentage of total species richness, the ratio between exotic and native species richness, exotic cover as a percentage of total cover, the ratio of exotic to native cover, and a weediness index. The weediness index was calculated using the following formula:

$$\text{weediness} = \frac{(\text{exotic/native species richness} + \text{exotic/native cover})}{2}$$

These condition variables were tested against the environmental, management and disturbance variables (independent variables) using two subsets of the data at the quadrat level: for all heath edge quadrats (n = 58) and for all heath centre quadrats (n = 50). Table 5.1 summarises the number of sites occurring for each of the categorical variables and Table 5.2 summarises the mean, standard deviation and range for the continuous variables.

Although both total species richness and total cover where tested, in most cases a significant difference in the total richness or cover was due to a difference in the native richness or cover. Therefore, in the following presentation and discussion of results, only native richness and cover are reported as they are more useful when assessing condition. An exception is made for total and native cover in the edge effect ratios, which responded quite differently and therefore both are reported. Other variables that had similar responses were the exotic/native richness ratio and exotic species richness as a percentage of total species richness (for both edge and centre), so the exotic/native ratio is used only. The results for exotic cover, exotic cover as a percent of total cover and the exotic/native cover ratio were also virtually identical, so exotic cover was considered the most useful. The vegetation structural variables, dominant and subdominant species and cover, and basal tree area were not analysed as there was either not enough variation or too few data points for analysis.

Table 5.2. Characteristics of continuous dependent variables and continuous independent variables for 50 heath–pasture boundaries surveyed in coastal heath in Tasmania

Variable	mean ± 1 st dev	range
Dependent variables		
total species richness	19.9 ± 0.4	4–38
native species richness	16.5 ± 0.4	1–33
exotic species richness	3.4 ± 0.2	0–12
exotic/native species richness	0.5 ± 0.1	0–10
total cover (cm)	1323.6 ± 24.8	32–3080
native cover (cm)	1144.1 ± 33.2	0–3080
exotic cover (cm)	179.5 ± 16.4	0–1405
exotic/native cover (cm)	1.6 ± 0.5	0–112.4
weediness index	1 ± 0.3	0–57.2
depth of penetration (m)	14.5 ± 1	0–19.5
richness penetration index	4.5 ± 0.3	0–9.5
cover penetration index	2.6 ± 0.3	0.5–11.5
weediness penetration index	3.8 ± 0.4	0–11.1
Independent variables		
Environmental and soil variables		
altitude (m)	29.3 ± 3.1	5–120
slope (degrees)	2.7 ± 0.6	0–26
edge soil hue	5.3 ± 0.1	3–8
edge soil value	3 ± 0.2	1.7–6
edge soil pH	5.9 ± 0.1	4.5–7.25
edge soil phosphorus (ppm)	9.3 ± 1.3	0.5–43
edge soil nitrogen (%)	0.3 ± 0.02	0.05–0.71
edge soil P/N (ppm/%)	44.1 ± 7.9	1.14–266.67
centre soil hue	5 ± 0.2	2–6
centre soil value	2.7 ± 0.2	1.7–6
centre soil pH	5.9 ± 0.1	4.5–7
edge soil phosphorus (ppm)	4.5 ± 0.5	0.5–16
edge soil nitrogen (%)	0.3 ± 0.02	0.03–0.69
centre soil P/N (ppm/%)	19.8 ± 2.3	2.5–92.3
edge/centre soil hue	1.1 ± 0.1	0.5–2.5
edge/centre soil value	1.2 ± 0.1	0.5–2.5
edge/centre soil pH	1 ± 0.01	0.8–1.2
edge/centre phosphorus (ppm)	5.2 ± 1.4	0.17–35
edge/centre nitrogen (%)	1.3 ± 0.1	0.3–3.5
Management and disturbance		
native grazing index	5.1 +/- 3	0–9
amount of super (kg of phosphorus/ha)	170.1 ± 19.3	0–490
edge/centre disturbance index	1.3 ± 0.1	0.3–4

Lifeform variables

Plant cover was further divided into 13 lifeform and other cover variables that were used to investigate how different plant lifeform groups responded to the environmental, management and disturbance variables. The seven native lifeform groups are fern, grass, graminoid, herb, shrub, native grass as a percent of total grass and native herb as a percent of total herb. Although native tree and *Xanthorrhoea australis* were measured as separate categories, there was not enough cover in enough quadrats to be used for analysis. There are four exotic lifeform groups: grass, herb, exotic grass as a percent of total grass and exotic herb as a percent of total herb; there was not enough cover for exotic shrubs or exotic trees for analysis. The two 'other' cover variables are bare ground and scat cover. However, there were virtually no scats present in the heath centre. The variables were again tested using two subsets of the data at the quadrat level: for all heath edge quadrats (n = 58) and for all heath centre quadrats (n = 50).

Edge effect and invasibility variables

Another ten dependent variables are used to assess edge effects and the invasibility of the heathland. Edge effect variables are the edge/middle ratios of total species richness, native species richness, exotic species richness, total plant cover, native plant cover and exotic plant cover.

High edge/middle ratios of total and native species richness indicate a prevalence of species at the edge compared with the centre of each of the vegetation types. This could indicate that conditions at the edge are more favourable than in the interiors, or that there is a substantial presence of edge exploiter species. Low ratios imply no such difference in conditions between edge and centre. High edge/middle total and native cover ratios indicate a sharp boundary and/or increased growth at the edge, a possible response to raised nutrient levels or water. Lower cover ratios can be due to a more diffuse boundary, with heath species invading the pasture or disturbance at the edge.

The exotic edge/middle species richness and cover ratios give some indication as to the invasibility of the heath. High ratios actually indicate lower exotic penetration, i.e. many species are at the edge of the heath and fewer are in the centre. A lower ratio indicates either higher exotic penetration or simply less exotic species at the heath edge.

The four invasibility variables are the depth of penetration of exotics into the heath, a penetration index for exotic species richness, a penetration index for exotic species cover and a penetration index for 'weediness'.

The depth value was simply the midpoint of the last quadrat in which an exotic species was found, expressed as distance in metres from the boundary. The exotic species richness penetration index was calculated for each transect in the following way:

$$= \frac{\Sigma \text{ (exotic species richness of quadrat x distance from boundary)}}{\Sigma \text{ (exotic species richness of quadrat)}}$$

The exotic species cover and weediness penetration indices were calculated by replacing exotic species richness with exotic species cover and the weediness index in the above formula, respectively. With these three indices, a higher value means both more exotic species (or exotic cover or weediness) and a deeper penetration into the heath.

These ten variables were tested against the environmental, management and disturbance variables at the transect level (n = 50). Where independent variables were collected at the quadrat level (all disturbance variables, soil nutrients, colour and texture and the *Phytophthora cinnamomi* score), they were grouped and tested according to quadrat position (e.g. heath edge quadrats, B quadrats, C quadrats).

For the native mammal trail analysis, the dependent variables were simply total, native and exotic species richness. The independent variables consisted of the 'treatment' (trail or no trail), presence or absence of bare ground, and distance from the heath–pasture boundary.

The relationships between the dependent and independent variables were tested using one-way ANOVAs where the independent variables were class data. Pearson's Product Moment Correlation Coefficient was used where the independent variables consisted of continuous data. Pairwise differences within the ANOVA were tested using Tukey's method. The Anderson-Darling statistic was used to test the residuals for normality for each ANOVA. Data were transformed where appropriate, using natural log, square root or arcsine transformations, or the lambda value suggested by a Box-Cox transformation, depending on the type and shape of the data (Sokal and Rohlf 1995). The Box-Cox transformation estimated a lambda value that minimised the standard deviation of the transformed variable. During transformation, zero values were dealt with by adding 1 to all values if log transforming or using the Box-Cox transformation, or adding 0.5 to all values if using a square root transformation (Sokal and Rohlf 1995). The transformation with the most normal residual was then used, meaning different variables may have had different transformations.

Some variables were not normal even after repeated attempts at transformation (for example exotic species richness in the heath centre quadrats, for which there were many zeros), and for these the Kruskal-Wallis rank order test was used as a non-parametric alternative to ANOVA. The native mammal trail variables also had to be treated in this manner. The Chi-squared test was used to test for association between bare ground and

mammal trails. Results were considered significant where $P < 0.05$. The analyses were done using the statistical package MINITAB (Release 12.2).

RESULTS: FACTORS AFFECTING HEATHLAND CONDITION AND WEED INVASION

General floristics

There were 322 vascular plant species found across the 313 quadrats. Of these, 268 were native and 54 were exotic species. The most common native species was *Leptospermum scoparium* (Manuka or Teatree) in 64% of quadrats, and the most common exotic species was *Hypochoeris radicata* (Cat's Ear) in 52% of quadrats. Of the three major types of heathland in Tasmania, wet heath and dry heath communities were sampled, but not rock pavement heath (see Kirkpatrick and Harris (1999b) for a complete classification of Tasmanian heathland communities). Seven species were considered rare or threatened in Tasmania. A full species list is given in Appendix 4 and shows both exotic and rare or threatened species.

Pasture quadrats had a significantly lower total and native species richness when compared with all the heath quadrats (Table 5.3, Figure 5.3a). However, the numbers of native species were actually higher, on average, than the number of exotic species in the pasture, showing significant reinvasion by the heath into pasture. Exotic species declined rapidly with distance from the heath boundary. Those quadrats closer to the boundary were generally significantly different to those further away from the boundary. Not surprisingly, the ratio of exotic to native species was significantly higher in pasture quadrats compared with all the heath quadrats.

Table 5.3. Significance values for total, native and exotic species richness and cover vs quadrat position. P, pasture quadrat; A, heath edge quadrat; B, quadrat 2–3 m from boundary; C, quadrat 5–6 m from boundary; D, quadrat 10–11 m from boundary; E, heath centre quadrat

condition variable	F	P	pairwise differences
total species richness	6.96	0.000	P<A, P<B, P<C, P<D, P<E
native species richness	13.88	0.000	P<A, P<B, P<C, P<D, P<E
exotic species richness	19.12	0.000	P>B, P>C, P>D, P>E, A>C, A>D, A>E, B>D, B>E
total cover	18.01	0.000	P<B, P<C, P<D, P<E, A<B, A<C, A<D, A<E
native cover	35.01	0.000	P<A, P<B, P<C, P<D, P<E, A<B, A<C, A<D, A<E
exotic cover	21.04	0.000	P>A, P>B, P>C, P>D, P>E, A>C, A>D, A>E, B>E

Total cover and native cover increased with distance into the heath (Figure 5.3b), whereas exotic cover dropped dramatically, becoming negligible from about 5 m onwards. Total cover was significantly lower in pasture quadrats compared with all the heath quadrats,

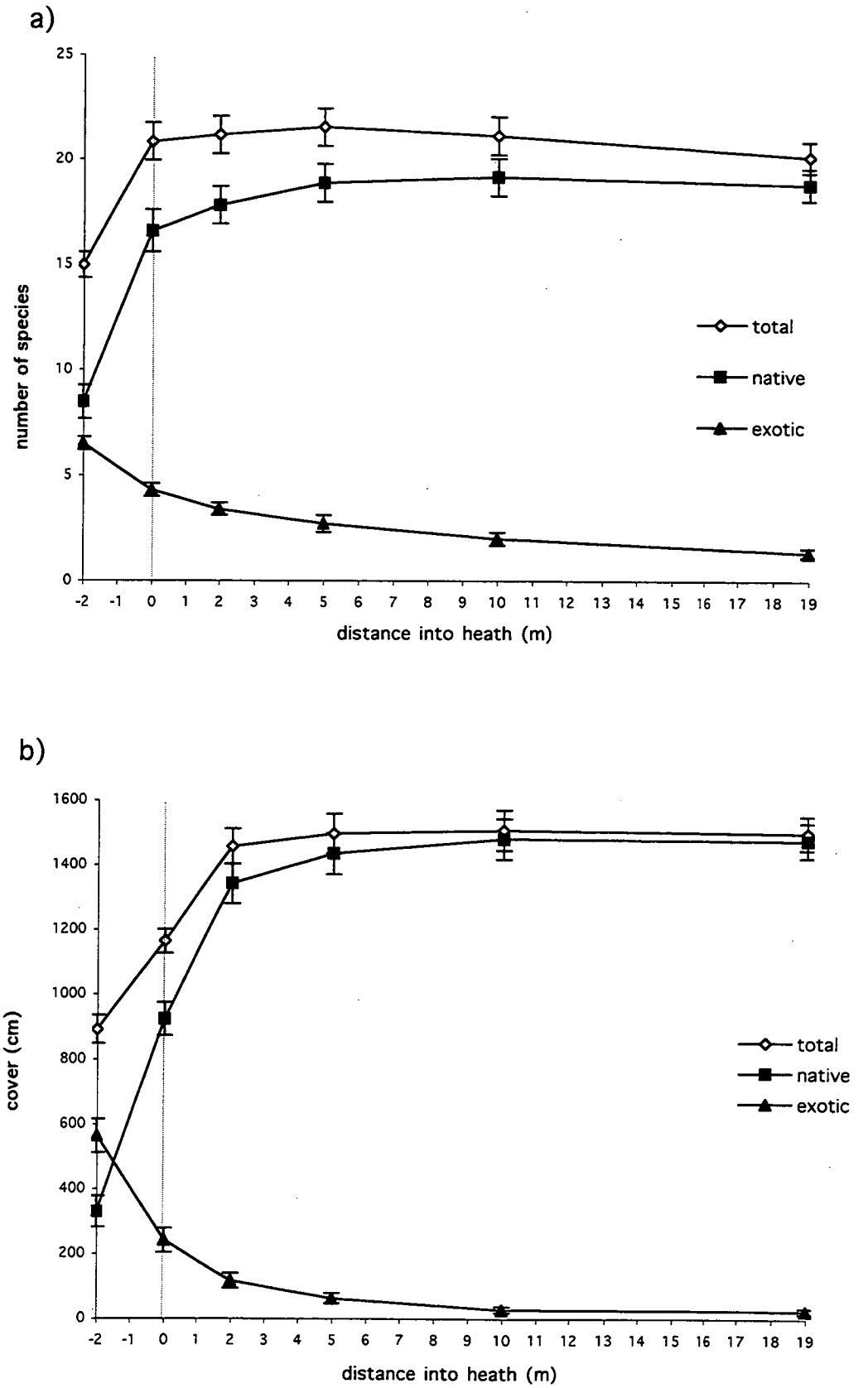


Figure 5.3. Total, native and exotic a) average species richness, and b) average cover over distance from heath edge. Standard error bars are shown. Heath–pasture boundary indicated by vertical line

and was also significantly lower at the heath edge compared with all other heath quadrats (Table 5.3). The amount of total cover in pasture and the heath edge quadrats was statistically the same. Native cover was also significantly lower in pasture than in heath and at heath edges than heath interiors. Exotic cover was always higher in the pasture quadrats and, as with species richness, those heath quadrats closer to the boundary generally had significantly more exotics than those further away from the boundary.

The ratio of edge/middle total species richness was positively correlated with the ratio of edge/middle native species richness (Table 5.4). So any effects on total species richness were usually due to effects on native species richness. The same was true for cover, with the addition that the total cover ratio was negatively correlated with the exotic cover ratio. In other words, a response from total cover was usually due to native cover, whereas exotic cover was showing the opposite tendency.

Total cover and total species ratios were inversely related (Table 5.4), i.e. when species richness was higher at the heath edge than the centre, cover was lower at the heath edge than the centre. This is common in heathland communities where there is often a large number of different species present, with no one species gaining dominance (Specht 1981a; Adam *et al.* 1990).

The exotic species richness edge/centre ratio was negatively correlated with the native cover edge/centre ratio (Table 5.4). Similarly, the exotic cover ratio was negatively correlated with the native cover ratio. In the field this would mean that when native cover in the centre was high compared with the edge, exotic species richness and cover were high at the edge compared with the centre. However, this does not imply any causal relationship. The two exotic ratios, richness and cover, were positively correlated. So an increase in the number of exotic species was associated with an increase in cover.

Table 5.4. Summary of correlation results for edge/centre ratios of total, native and exotic cover and richness variables. First figure is Pearson's correlation and second figure is P value. +, positive correlation; -, negative correlation; ns, not significant

	total cover	native cover	exotic cover	total richness	native richness
native cover	+0.871 0.000	—			
exotic cover	ns	-0.467 0.000	—		
total richness	-0.317 0.021	ns	ns	—	
native richness	ns	ns	ns	+0.918 0.000	—
exotic richness	ns	-0.291 0.034	+0.471 0.000	ns	ns

The condition variables generally responded to the same degree at both edge and centre. The exception was exotic species richness (and exotic richness % and exotic/native richness) which responded to the management variables much less in the heath centre.

This was probably due to the lower presence of exotics further into the heath. Native grass was the most responsive of the lifeform variables, at both the edge and the centre (Figure 5.4). Native grass as a percentage of total grass was affected by many management variables in the heath centre but none at the heath edge. Scat cover at the edge and bare ground at the heath centre also responded to many independent variables. There were more significant results for exotic herb cover than for exotic grass cover at the heath edge, but the reverse for the heath centre.

In the following description and discussion of results, a few short-hands have been used to simplify the text. 'Species richness' is shortened to just 'richness'; 'native species richness as a percentage of total species richness' is shortened to 'native richness %' (the same for exotic). For the lifeform cover variables, 'native grass %' means 'native grass cover as a percentage of total grass cover' not 'native grass cover as a percentage of total cover' (which was not used). The same holds for 'native herb %', 'exotic grass %' and 'exotic herb %'. To reduce the number of figures, exotic/native richness for edge and centre, native grass % and herb % for edge and centre, and exotic grass % and herb % for edge and centre are not presented.

Environment

Region was a significant indicator of variation for the following variables (Figure 5.5a–f). For condition at the edge: native richness, exotic richness, exotic/native richness, native cover, exotic cover, weediness; and native richness for condition at the centre. For lifeform at the edge: scat, native shrub, native herb %, exotic grass, exotic herb, exotic herb %; for lifeform at the centre: fern, native grass, native herb %. For edge effect ratios: edge/centre total richness, edge/centre native richness, edge/centre native cover. There was no regional variation for the four invasibility variables.

For the condition variables, there were many more regional differences for species richness and cover at the heath edge than there were at the heath centre. This means that the regional differences in the edge/centre ratio are more likely to be due to differences at the edges (such as pasture variables) rather than at the centres. This was also the case for the lifeform cover variables, although there was no significant regional variation in any of the exotic lifeform groups in the heath centre.

The regional differences can be summarised as an overall difference between the Flinders Island, Waterhouse and Northeast regions (in the northeast corner of the state) compared with the East Coast, Northwest and Southeast regions (all other parts of the state surveyed). Table 5.5 shows the characteristics of these two 'super regions'. Although the summary holds true for most transects, there were four variables for which there were still regional variations when split into the two super regions. These were native species

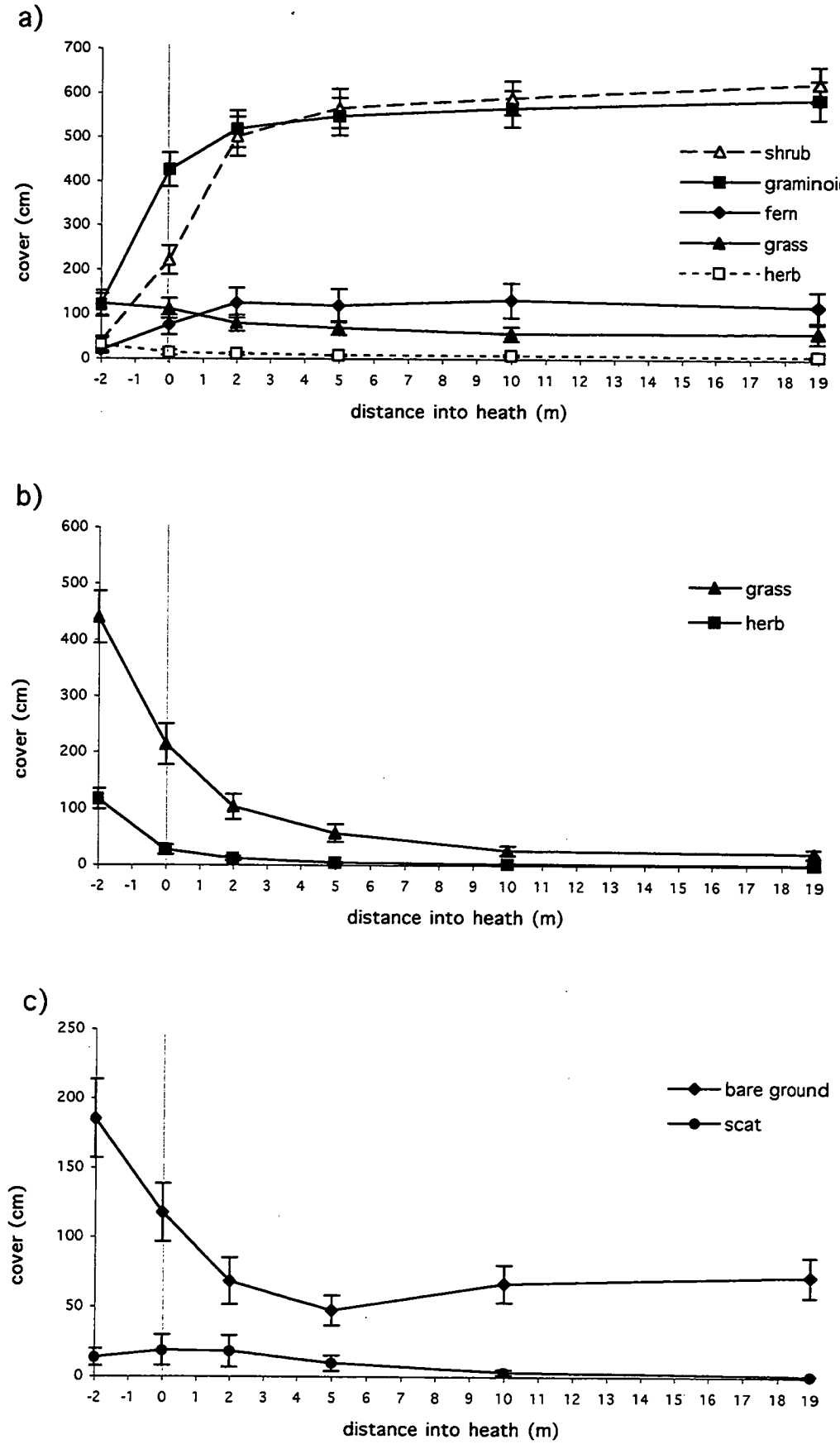


Figure 5.4. Average a) native lifeform cover, b) exotic lifeform cover and c) miscellaneous cover over distance from heath edge. Standard error bars are shown. Heath–pasture boundary indicated by vertical line

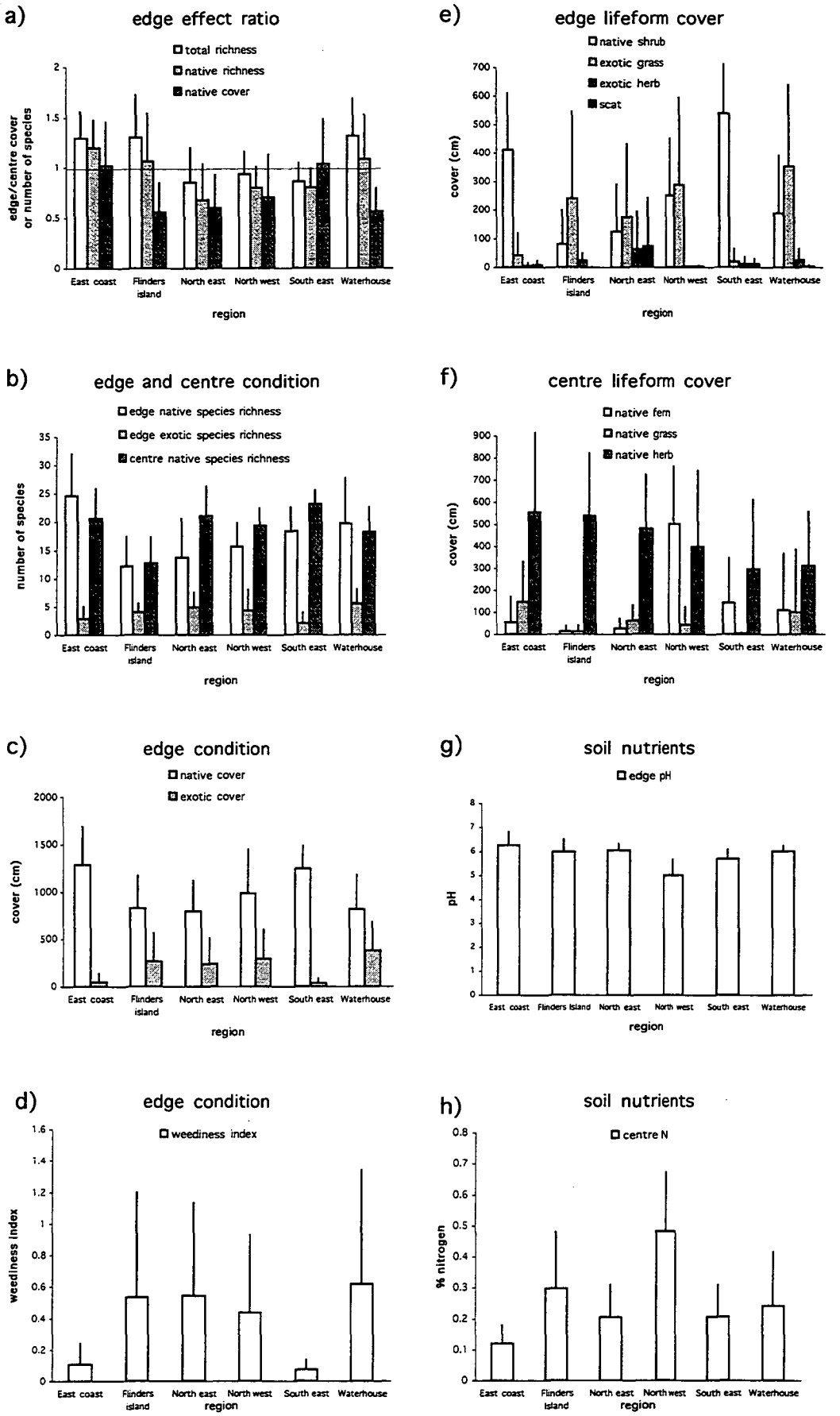


Figure 5.5. Regional differences in a) edge effect ratios, b–d) edge and centre condition, e) edge lifeform cover and f) centre lifeform cover. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness index

richness at the edge and at the centre, scat cover at the edge and native grass cover at the centre. In order to get around this persistent regional variation, the regions were combined and split in a different way to the super regions described above, for these four variables. For native richness at the edge, East Coast and Waterhouse had higher richness and Flinders, Northeast, Northwest and Southeast had lower richness. For native richness at the centre, Flinders Island was particularly low and was omitted. For scat cover at the edge, Southeast and Northeast were particularly high and were omitted (there were two few sites for these two regions to be tested together as a super region). Finally, for native grass in the centre, East Coast was much higher than other regions and was omitted. All other variables that had significant regional differences were re-tested for the two original super regions separately. Soil variables were also re-tested using all versions of the super regions.

Table 5.5. Regional differences in species richness, cover, lifeform cover and soil variables for the Northeast and Other super regions

	'NORTHEAST'	'OTHER'
	Flinders Is, Waterhouse, Northeast	**East coast, Northwest, Southeast**
species richness	low natives at edge	med/high natives at edge
	high exotics at edge	low/med exotics at edge
	high native edge/centre ratio	low native edge/centre ratio
cover	low native cover at edge	high native cover at edge
	med/high exotic cover at edge	very low exotic cover at edge
	low native edge/centre ratio	high native edge/centre ratio
lifeform cover	low/med native shrub at edge	variable native shrub at edge
	med/high exotic grass at edge	low/med exotic grass at edge
	low exotic herb at edge	high exotic herb at edge
	med/high exotic herb % at edge	low exotic herb % at edge
	low fern in heath centre	variable fern in heath centre
	low native herb % at heath centre	high native herb % at heath centre
soil nutrients	high pH at edge	low/med pH at edge
	medium nitrogen levels at centre	low/high nitrogen levels at centre

Aspect, alignment, geology, slope, landform and altitude had some effect on condition variables, edge effect variables and exotic penetration indices (Table 5.6). Significant differences in edge effect ratios were seen for different aspects. However, these results can be ignored for the following reasons. Warmer and drier aspects (NW, N, W), which had lower total richness and higher native cover edge effect ratios, were found more often in the Other region (Figure 5.6a). There were no differences in these variables between aspects when the regions were tested separately. Therefore, regional variation would explain the differences found. Exotic cover did not vary by region, and was significantly lower in SE aspects (Figure 5.6b). However, there were only 6 quadrats with this aspect

Table 5.6. Summary of significant parametric and non-parametric results for environmental variables

VARIABLES	DF	penetration index exotic depth[1] H	P	penetration index exotic richness F	P	penetration index exotic cover F	P	penetration index exotic weediness F	P	edge effect ratio total richness F	P	edge effect ratio native richness F	P	edge effect ratio exotic richness F	P	edge effect ratio total cover F	P
region	5									4.78	0.001	2.76	0.028				
aspect	3									2.97	0.040						
alignment	2																
geology	2									5.23	0.020						
landform	2	6.98	0.031													4.87	0.011
altitude														0.293	+ 0.033		
slope class	3													3.44	0.023		
average slope																	

VARIABLES	DF	edge effect ratio native cover F	P	edge effect ratio exotic cover F	P	edge condition native richness F	P	edge condition exotic richness F	P	edge condition ex:nat richness F	P	edge condition native cover F	P	edge condition exotic cover F	P	edge condition weediness index F	P
region	5	3.31	0.011			3.97	0.004	2.61	0.035	2.68	0.032	12.23[1]	0.032	2.84	0.024	2.62	0.035
aspect	3	3.85	0.014	5.66	0.002												
alignment	2					4.13	0.021										
geology	2											3.18	0.049				
landform	2	3.40	0.041														
altitude																	
slope class	3					3.21	0.030										
average slope																	

VARIABLES	DF	centre condition native richness F	P	centre condition exotic richness F	P	centre condition ex:nat richness F	P	centre condition native cover F	P	centre condition exotic cover F	P	centre condition weediness index F	P	edge lifeform cover scat[1] F	P	edge lifeform cover bare ground F	P
region	5	5.96	0.000											20.43	0.001		
aspect	3							3.85	0.015								
alignment	2																
geology	2	6.73	0.003											6.85	0.032		
landform	2	6.05	0.005														
altitude																	
slope class	3																
average slope																	

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.6. Summary of significant parametric and non-parametric results for environmental variables

		edge lifeform cover native fern[1]		edge lifeform cover native grass[1]		edge lifeform cover native grass %		edge lifeform cover native gram		edge lifeform cover native herb[1]		edge lifeform cover native herb %		edge lifeform cover native shrub		edge lifeform cover exotic grass	
		F	P	F	P	F	P	F	P	F	P	F	P	F	P	F	P
region	5											12.08[1]	0.034	4.53	0.002	3.28	0.012
aspect	3																
alignment	2																
geology	2									11.52	0.003	11.60[1]	0.003				
landform	2	6.15	0.046	6.39	0.041												
altitude																	
slope class	3																
average slope												0.291	+ 0.027				

		edge lifeform cover exotic grass %		edge lifeform cover exotic herb		edge lifeform cover exotic herb %[1]		centre lifeform cover scat		centre lifeform cover bare ground		centre lifeform cover native fern		centre lifeform cover native grass[1]		centre lifeform cover native grass %	
		F	P	F	P	F	P	F	P	F	P	F	P	F	P	F	P
region	5			3.89	0.004	23.82	0.000					11.57[1]	0.041	14.05	0.015		
aspect	3									5.31	0.003						
alignment	2																
geology	2					9.39	0.009					6.21[1]	0.045				
landform	2																
altitude																	
slope class	3																
average slope												0.547	+ 0.000				

		centre lifeform cover native gram		centre lifeform cover native herb[1]		centre lifeform cover native herb %[1]		centre lifeform cover native shrub		centre lifeform cover exotic grass		centre lifeform cover exotic grass %		centre lifeform cover exotic herb		centre lifeform cover exotic herb %	
		F	P	F	P	F	P	F	P	F	P	F	P	F	P	F	P
region	5			12.16	0.033	12.16	0.033										
aspect	3	3.04	0.038														
alignment	2																
geology	2			6.88	0.032	6.88	0.032										
landform	2																
altitude																	
slope class	3																
average slope																	

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

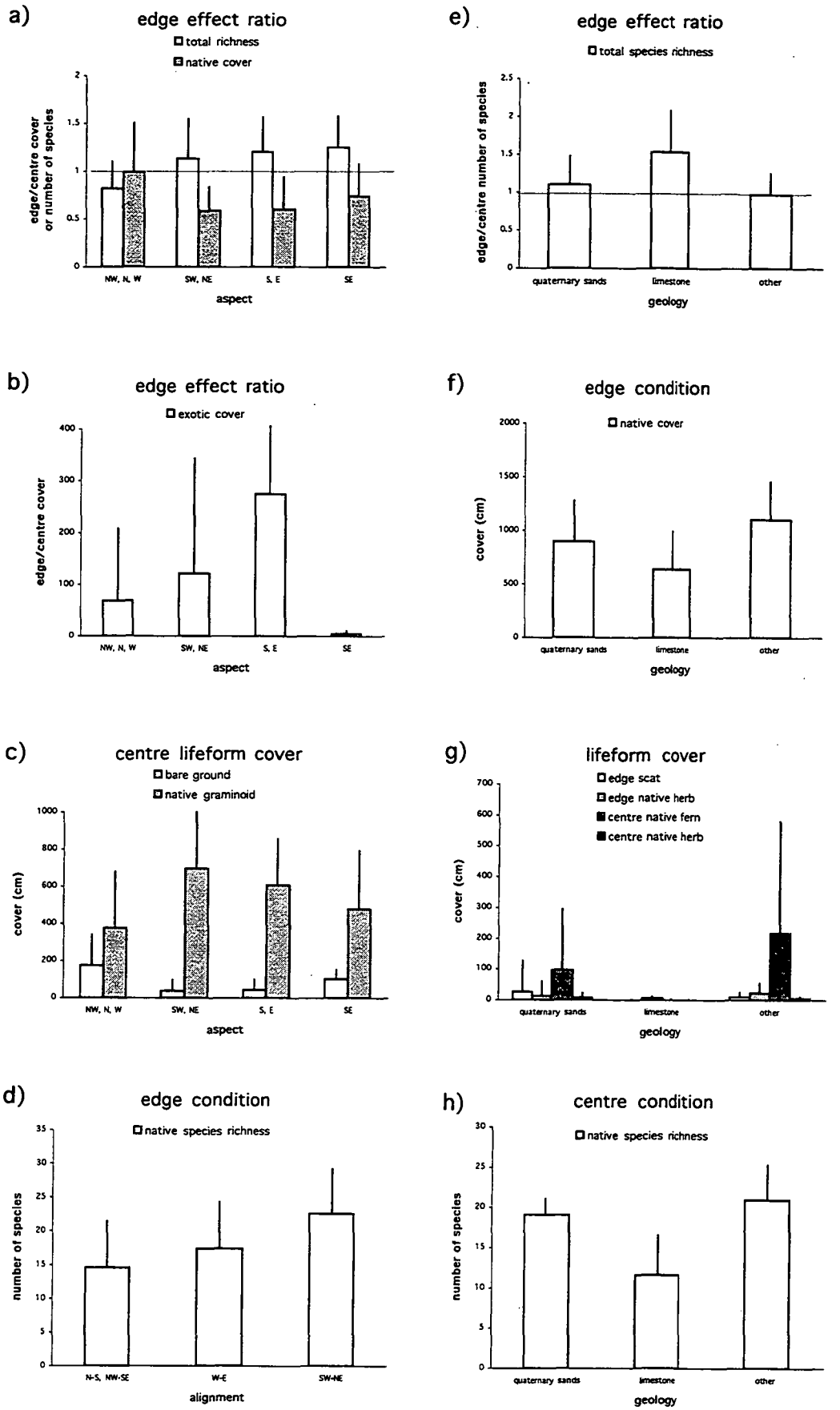


Figure 5.6. Differences in a–c) edge effect ratio and centre lifeform cover for the aspect of the heath–pasture boundary, d) edge condition for heath–pasture boundary alignment, and e–h) edge effect ratio, edge and centre condition and lifeform cover for geology. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation.

and thus it was combined with the S and E aspects after which there was no significant difference in exotic cover. For the centre lifeform cover variables bare ground and native graminoid cover, the significant differences observed could be due to aspect because there was no variation by region. Heathland centres with warmer and drier aspects had more bare ground and less native graminoid cover (Figure 5.6c).

The alignment of the boundary was not a significant factor for invasibility indices or edge effect variables but it was for native species richness at the heath edge (Figure 5.6d). The alignment of the boundary that would receive the most sun (aligned along the SW–NE axis) had significantly higher native richness at the edge than the alignment with the least sun (NW–SE). This result was for the entire dataset and the Other region but not for the Northeast.

Geology, like aspect, showed some regional variation, with heathland on limestone only being found on Flinders Island in the Northeast region. Therefore, the higher total species richness edge effect ratio, lower edge native cover, and lower centre native fern (Figure 5.6e–g) on limestone (all variables that differ regionally) could be due to some feature of the environment or management specific to these sites on Flinders Island. However, it is still possible that the geology is influencing the vegetation instead of or as well as the regional difference. Five variables that did not differ between regions did still differ between geologies. Edge scat cover, edge native herb cover, centre native herb cover and centre native species richness (Figure 5.6g–h) were all lower on limestone, and exotic herb % cover was higher on limestone. When the Northeast region was tested separately, fern cover and native herb % cover in the heath centre were low for Quaternary sands and limestone and high for other geologies. Also in the Northeast, phosphorus at the heath edge was actually higher for Quaternary sands compared with other geologies — perhaps reflecting the greater quantity of fertiliser required by these usually low-nutrient sandy soils. In summary, exotic herbs were higher and native cover and richness were lower on limestone, either due to the geology or to some other aspect of the region.

The range in slopes in the survey was very small, with most slopes being less than 6°. The two main variables that responded to slope were edge/centre exotic richness and edge native richness. Edge/centre exotic richness was significantly higher for relatively steeper slopes (Figure 5.7a), although in all slopes the number of exotic species was far higher at the edge of the heath. The lower ratio for flatter slopes implies there were either less species at the edge or more exotic species getting into the heath centre. Native richness at the heath edge generally increased with increasing slope (Figure 5.7b). However, this variable was subject to regional differences and the slope categories of 2° and 3–5° were found all or mostly in the Northeast region. The regional tests showed edge native cover and native herb % cover increased and edge exotic herb % decreased with increasing slope in the Northeast region.

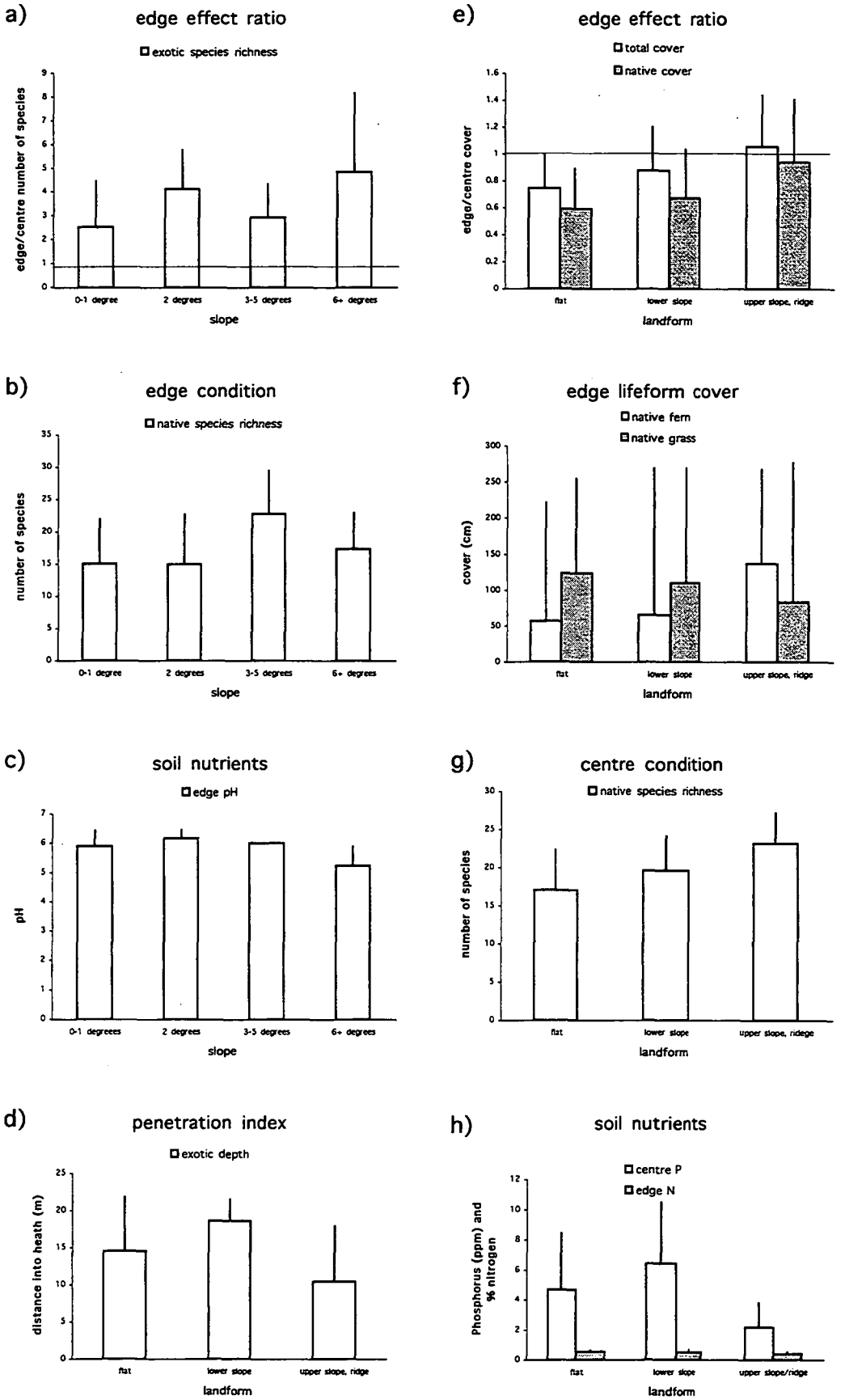


Figure 5.7. Differences in a–c) edge effect ratio, native species richness and soil nutrients for the slope of the heath–pasture boundary, and d–f) penetration index, edge effect ratio, edge lifeform cover, centre native species richness and soil nutrients for the landform on which the heath–pasture boundary was found. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of penetration index

Although these variables appeared to be responding to slope, there were some relationships between slope and soil nutrient variables which could be influencing the vegetation responses. Less steep slopes had higher pH at both the edge and the centre and lower nitrogen at the centre (Figure 5.7c). In the Northeast region, phosphorus at the centre was higher for flatter slopes. Thus the heath edges of flatter sites had lower native richness and less native herb cover on more alkaline soils. The centre of flatter sites was also more alkaline, with lower nitrogen levels and fern cover but relatively higher numbers of exotic species. Flatter sites in the northeast had less native cover and native herb cover but more exotic herb cover and higher phosphorus levels. To generalise, the results show an association between alkaline soils with higher phosphorus levels and the higher presence of exotics.

Landform affected weed invasion and the edge effect and condition variables. Lower slopes showed a greater depth of weed penetration (Figure 5.7d), and flat areas experienced less total and native cover at the edge than the centre, lower native fern cover and higher native grass cover at the edge, lower native species richness in the centre (Figure 5.7e–g), and lower native fern cover in the centre in the Northeast region. Native cover is the only one of these variables that was subject to regional variation but it can be used in this case because the different landform categories were evenly spread across the two regions. Flat areas probably experience more runoff and deposition that could bring in nutrients from surrounding farmlands. Testing the soil nutrient variables against landform did show that flat areas had higher nitrogen at the edge. In the Northeast region, flat areas had higher edge and centre phosphorus and nitrogen. Higher phosphorus at the centre was also found for lower slopes, for the whole data set.

There was a positive correlation between altitude and the edge/middle ratio of exotic species richness, meaning more weeds penetrated into the heath centre in lower altitudes (Table 5.6). It is unlikely that altitude was the real influence in this case because, firstly, the altitude range was quite small (5–120 m) and, secondly, there was a strongly negative correlation between pH at the heath edge and altitude that is more likely to explain weed penetration. At lower altitudes the soils were more alkaline at the heath edge, and this relationship probably had more influence on weed penetration than altitude alone.

Soil and nutrients

Nitrogen and phosphorus decreased from the pasture into the heath centre (Figure 5.8). Correlations between edge and centre nitrogen and phosphorus show that, as phosphorus increased, nitrogen increased, at both the edge and the centre, and between the edge and centre (Table 5.7). The two ratios of edge/middle phosphorus and edge/middle nitrogen were also positively correlated with each other. However, just because phosphorus might

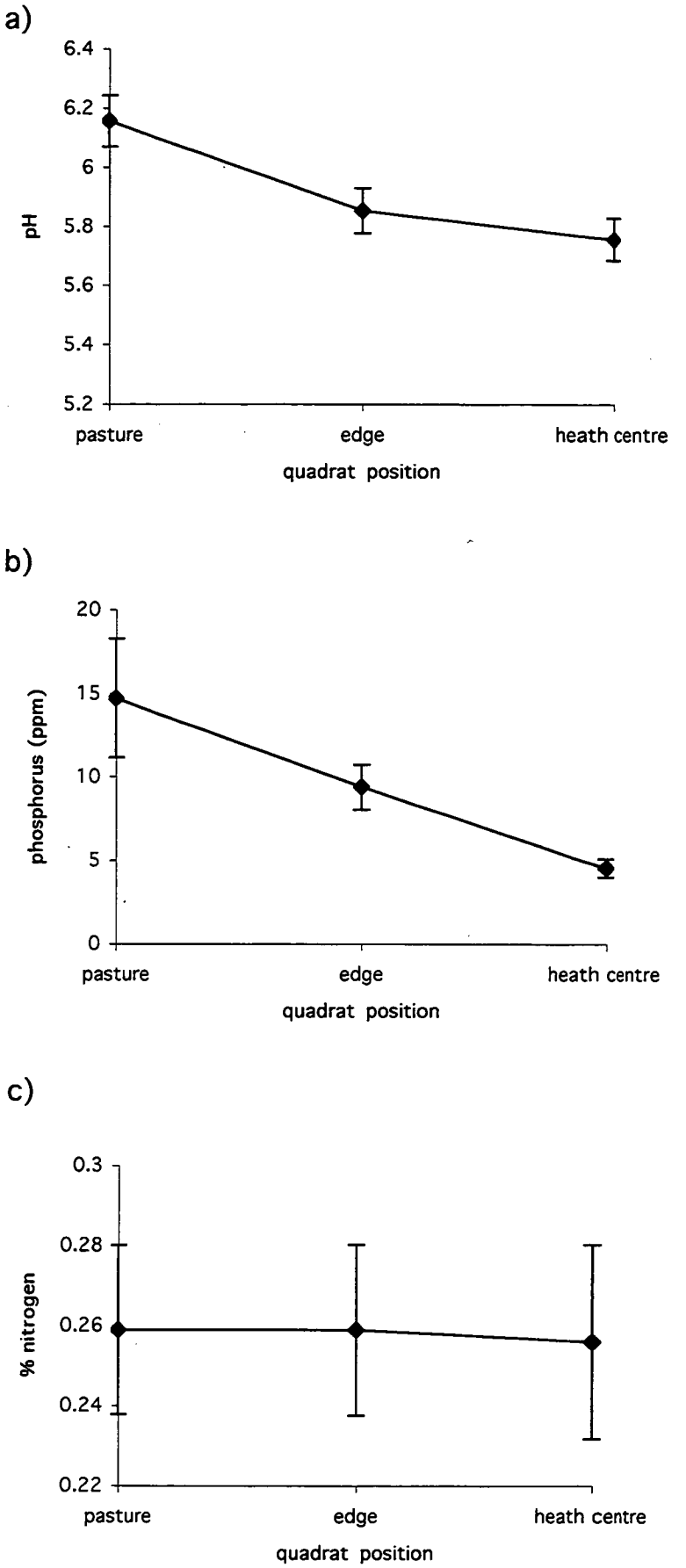


Figure 5.8. Soil a) pH, b) phosphorus levels, and c) nitrogen levels in the pasture, heath edge and heath centre. Standard error bars are shown

be high at the edge did not mean it was high at the centre, and the same held for nitrogen. This shows that any soil nutrient increase resulting from the management of the pasture may only affect the heathland edge and not penetrate too deeply beyond that.

Table 5.7. Summary of correlation results between all edge and centre soil pH, phosphorus and nitrogen variables, and between edge/centre phosphorus and edge/centre nitrogen. First figure is Pearson's correlation and second figure is P value. +, positive correlation; -, negative correlation; ns, not significant; n/a, not applicable

	edge pH	edge P	edge N	centre pH	centre P	edge/ centre P
edge P	ns	—				
edge N	ns	+0.364 0.009	—			
centre pH	+0.683 0.000	ns	ns	—		
centre P	ns	ns	+0.344 0.021	ns	—	
centre N	ns	ns	ns	ns	+0.615 0.000	—
edge/ centre N	n/a	n/a	n/a	n/a	n/a	+0.341 0.014

Neither edge effect ratio for soil variables had a significant relationship with the ratio of edge/middle exotic species richness (Table 5.8). When the ratio of phosphorus to nitrogen was tested, there was a weak positive correlation with the phosphorus/nitrogen ratio at the edge of the heath and the edge/centre total cover ratio. That is, more phosphorus than nitrogen at the heath edge meant more cover at the heath edge. There were no correlations between the phosphorus/nitrogen ratio at the heath centre and the species richness and cover ratios.

Soil phosphorus (Figure 5.8a) did not have any effect on the penetration indices or the edge/centre cover or richness ratios but did appear to affect condition and lifeform variables. At the heath edge, phosphorus levels were positively correlated with exotic richness %, exotic/native richness, exotic cover, exotic cover %, exotic/native cover, the weediness index, scat cover, exotic grass cover and exotic grass % (Table 5.8). Phosphorus levels were negatively correlated with native richness. At the heath centre, phosphorus was positively correlated with exotic richness % and exotic/native cover and negatively correlated with native richness. In general, higher levels of phosphorus seemed to result in higher richness and cover of weeds (particularly grasses), and more scats (implying more grazing). Higher phosphorus levels did not appear to favour native species.

Soil nitrogen had a similar effect on native and exotic richness and cover, generally increasing exotic richness and cover and decreasing native species richness. Figure 5.8b shows that nitrogen levels were very similar between pasture and heath edge quadrats. Changes in nitrogen levels at the edge were probably due to an increased presence of nitrogen-fixing exotics, as nitrogen was most usually supplied by sowing clovers and

Table 5.8. Summary of significant results for parametric and non-parametric soil nutrient variables

VARIABLES	penetration index exotic depth[1]			penetration index exotic richness			penetration index exotic cover			penetration index exotic weediness			edge effect ratio total richness			edge effect ratio native richness			edge effect ratio exotic richness			edge effect ratio total cover		
	H	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
edge soil hue class				**7.26**	**2**	**0.002**																**4.72**		**0.022**
edge soil value																						**0.285**	**+**	**0.041**
edge soil chroma																								
edge soil texture																								
edge soil pH	**0.283**	**+**	**0.049**																					
edge soil phosphorus																**0.293**	**-**	**0.033**						
edge soil nitrogen	**0.379**	**+**	**0.007**																					
edge soil P/N																						**0.323**	**+**	**0.018**
centre soil hue class	**13.90**	**2**	**0.001**	**7.93**	**2**	**0.001**				**5.99**	**2**	**0.005**												
centre soil value													**0.320**	**-**	**0.021**	**0.300**	**-**	**0.031**				**0.398**	**+**	**0.004**
centre soil chroma																								
centre soil texture																								
centre soil pH	**0.345**	**+**	**0.015**	**0.394**	**+**	**0.005**				**0.309**	**+**	**0.031**												
centre soil phosphorus																						**0.284**	**-**	**0.039**
centre soil nitrogen																								
centre soil P/N																								
edge/centre soil hue	**0.488**	**-**	**0.001**	**0.350**	**-**	**0.014**																**0.274**	**-**	**0.049**
edge/centre soil value				**0.285**	**-**	**0.047**																		
edge/centre soil pH																			**0.339**	**+**	**0.014**			
edge/centre phosphorus																								
edge/centre nitrogen																								

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.8. Summary of significant results for parametric and non-parametric soil nutrient variables

	edge effect ratio native cover			edge effect ratio exotic cover			edge condition native richness			edge condition exotic richness			edge condition ex:nat richness			edge condition native cover			edge condition exotic cover			edge condition weediness index		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
edge soil hue class																								
edge soil value																								
edge soil chroma							**4.81**	**1**	**0.033**															
edge soil texture																								
edge soil pH										**0.324**	**+**	**0.019**	**0.280**	**+**	**0.044**									
edge soil phosphorus							**0.387**	**-**	**0.004**				**0.315**	**+**	**0.021**				**0.305**	**+**	**0.027**	**0.364**	**+**	**0.007**
edge soil nitrogen							**0.325**	**-**	**0.020**				**0.332**	**+**	**0.017**				**0.348**	**+**	**0.012**	**0.373**	**+**	**0.007**
edge soil P/N																								
centre soil hue class	**3.42**	**2**	**0.040**				n/a			n/a			n/a			n/a			n/a			n/a		
centre soil value	**0.294**	**+**	**0.035**				n/a			n/a			n/a			n/a			n/a			n/a		
centre soil chroma							n/a			n/a			n/a			n/a			n/a			n/a		
centre soil texture							n/a			n/a			n/a			n/a			n/a			n/a		
centre soil pH							n/a			n/a			n/a			n/a			n/a			n/a		
centre soil phosphorus	**0.306**	**+**	**0.026**				n/a			n/a			n/a			n/a			n/a			n/a		
centre soil nitrogen							n/a			n/a			n/a			n/a			n/a			n/a		
centre soil P/N							n/a			n/a			n/a			n/a			n/a			n/a		
edge/centre soil hue							n/a			n/a			n/a			n/a			n/a			n/a		
edge/centre soil value							n/a			n/a			n/a			n/a			n/a			n/a		
edge/centre soil pH							n/a			n/a			n/a			n/a			n/a			n/a		
edge/centre phosphorus							n/a			n/a			n/a			n/a			n/a			n/a		
edge/centre nitrogen							n/a			n/a			n/a			n/a			n/a			n/a		

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.8. Summary of significant results for parametric and non-parametric soil nutrient variables

	centre condition native richness			centre condition exotic richness			centre condition ex:nat richness			centre condition native cover			centre condition exotic cover			centre condition weediness index			edge lifeform cover scat			edge lifeform cover bare ground		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
edge soil hue class	n/a			n/a			n/a			n/a			n/a			n/a								
edge soil value	n/a			n/a			n/a			n/a			n/a			n/a								
edge soil chroma	n/a			n/a			n/a			n/a			n/a			n/a								
edge soil texture	n/a			n/a			n/a			n/a			n/a			n/a			**14.62**[1]	**3**	**0.002**			
edge soil pH	n/a			n/a			n/a			n/a			n/a			n/a								
edge soil phosphorus	n/a			n/a			n/a			n/a			n/a			n/a			**0.414**	**+**	**0.002**			
edge soil nitrogen	n/a			n/a			n/a			n/a			n/a			n/a						**0.331**	**-**	**0.018**
edge soil P/N	n/a			n/a			n/a			n/a			n/a			n/a								
centre soil hue class				**15.63**[1]	**2**	**0.001**	**13.36**[1]	**2**	**0.001**	**5.01**	**2**	**0.011**	**15.46**[1]	**2**	**0.000**	**16.27**[1]	**2**	**0.000**	n/a			n/a		
centre soil value										**0.503**	**-**	**0.000**	**0.298**	**+**	**0.037**				n/a			n/a		
centre soil chroma																			n/a			n/a		
centre soil texture																			n/a			n/a		
centre soil pH							**0.322**	**+**	**0.024**				**0.357**	**+**	**0.012**	**0.405**	**+**	**0.004**	n/a			n/a		
centre soil phosphorus	**0.380**	**-**	**0.006**				**0.295**	**+**	**0.038**	**0.305**	**+**	**0.031**							n/a			n/a		
centre soil nitrogen	**0.288**	**-**	**0.043**																n/a			n/a		
centre soil P/N																			n/a			n/a		

	edge lifeform cover native fern			edge lifeform cover native grass[1]			edge lifeform cover native grass %			edge lifeform cover native gram			edge lifeform cover native herb			edge lifeform cover native herb %			edge lifeform cover native shrub			edge lifeform cover exotic grass		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
edge soil hue class				**13.23**	**2**	**0.001**																		
edge soil value																								
edge soil chroma																								
edge soil texture				**10.39**	**3**	**0.016**																		
edge soil pH																								
edge soil phosphorus							**0.405**	**-**	**0.003**													**0.294**	**+**	**0.032**
edge soil nitrogen																						**0.365**	**+**	**0.008**
edge soil P/N																								

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.8. Summary of significant results for parametric and non-parametric soil nutrient variables

	edge lifeform cover exotic grass %			edge lifeform cover exotic herb			edge lifeform cover exotic herb %			centre lifeform cover scat			centre lifeform cover bare ground			centre lifeform cover native fern			centre lifeform cover native grass[1]			centre lifeform cover native grass %		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
edge soil hue class	**6.82[1]**	**2**	**0.033**							n/a			n/a			n/a			n/a			n/a		
edge soil value										n/a			n/a			n/a			n/a			n/a		
edge soil chroma										n/a			n/a			n/a			n/a			n/a		
edge soil texture	**8.43[1]**	**3**	**0.038**							n/a			n/a			n/a			n/a			n/a		
edge soil pH							**0.296**	**+**	**0.033**	n/a			n/a			n/a			n/a			n/a		
edge soil phosphorus	**0.435**	**+**	**0.001**							n/a			n/a			n/a			n/a			n/a		
edge soil nitrogen										n/a			n/a			n/a			n/a			n/a		
edge soil P/N										n/a			n/a			n/a			n/a			n/a		
centre soil hue class	n/a			n/a			n/a												**9.62**	**2**	**0.008**			
centre soil value	n/a			n/a			n/a						**0.496**	**+**	**0.000**									
centre soil chroma	n/a			n/a			n/a															**3.94[1]**	**1**	**0.047**
centre soil texture	n/a			n/a			n/a												**8.74**	**3**	**0.033**			
centre soil pH	n/a			n/a			n/a									**0.405**	**-**	**0.004**				**0.326**	**+**	**0.022**
centre soil phosphorus	n/a			n/a			n/a																	
centre soil nitrogen	n/a			n/a			n/a						**0.290**	**-**	**0.041**									
centre soil P/N	n/a			n/a			n/a																	

	centre lifeform cover native gram			centre lifeform cover native herb[1]			centre lifeform cover native herb %[1]			centre lifeform cover native shrub			centre lifeform cover exotic grass			centre lifeform cover exotic grass %			centre lifeform cover exotic herb			centre lifeform cover exotic herb %		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
centre soil hue class				**7.22**	**2**	**0.027**	*	*	*				**13.62[1]**	**2**	**0.001**	**12.65[1]**	**2**	**0.002**	**8.15[1]**	**2**	**0.017**	**7.91[1]**	**2**	**0.019**
centre soil value	**0.346**	**-**	**0.015**										**0.299**	**+**	**0.037**							**0.342**	**+**	**0.016**
centre soil chroma																								
centre soil texture	**3.17**	**3**	**0.033**																					
centre soil pH													**0.347**	**+**	**0.015**				**0.395**	**+**	**0.005**	**0.342**	**+**	**0.016**
centre soil phosphorus																								
centre soil nitrogen																								
centre soil P/N																								

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

subclovers rather than directly as a fertiliser. Nitrogen at the heath edge was positively correlated with depth of exotic penetration, exotic richness %, exotic/native richness, exotic cover, exotic cover %, exotic/native cover, the weediness index and exotic grass cover (Table 5.7). It was negatively correlated with native species richness. Nitrogen at the heath centre was positively correlated with native cover but negatively correlated with native richness and bare ground.

Soil pH also decreased over distance into the heath (Figure 5.8c). The pH at the heath edge and centre were positively correlated with each other (Table 5.7), and both were positively correlated with the depth of exotic penetration (Table 5.8). In other words, when the pH became more alkaline weeds invaded deeper into the heathland. Also, the edge/middle ratio of soil pH was positively correlated with edge/middle ratio of exotic species richness. That is, where there was higher alkalinity at the edge than the middle there was a correspondingly higher number of exotics at the edge than the middle. Increasing alkalinity in the heath centre was also correlated with increasing penetration indices of exotic species richness and the weediness index.

There were many significant correlations between pH and the condition and lifeform variables (Table 5.8). The positive correlations between soil pH at the heath edge and exotic richness, exotic richness %, exotic/native richness and exotic herb cover % shows a strong association between alkalinity and exotic invasion. Soil pH in the heath centre was also positively correlated with exotic richness %, exotic/native richness, exotic cover, exotic cover %, exotic/native cover, the weediness index, exotic grass cover, exotic herb cover and exotic herb % cover. The only variable negatively correlated with centre pH was fern cover.

The colour (hue), lightness (value), saturation (chroma) and texture of the soil were related to the nutrient levels and pH of the soil. The positive correlation between soil hue and soil pH showed that the lower the hue (more reddish soils), the lower the pH, for both the edge and the centre; the higher the hue (more yellowish soils), the higher the pH. Reddish soils in the heath centre were also slightly more acidic in the Other region, and reddish soils at the heath edge had higher phosphorus in the both the Northeast and Other regions.

Hue appeared to affect weed penetration, and condition, edge effect and lifeform cover variables. A lower hue at the heath centre was associated with less deep weed penetration, lower penetration indices for exotic richness and weediness, fewer exotic species and less exotic cover in the centre, lower exotic/native richness and cover ratios in the centre, a lower weediness index in the centre, and less exotic grasses and herbs in the centre (Figure 5.9a–e). There was also a lower ratio of edge/centre native cover (Figure 5.9f), more native cover in the centre (Figure 5.9c), but less native grass and herb cover in

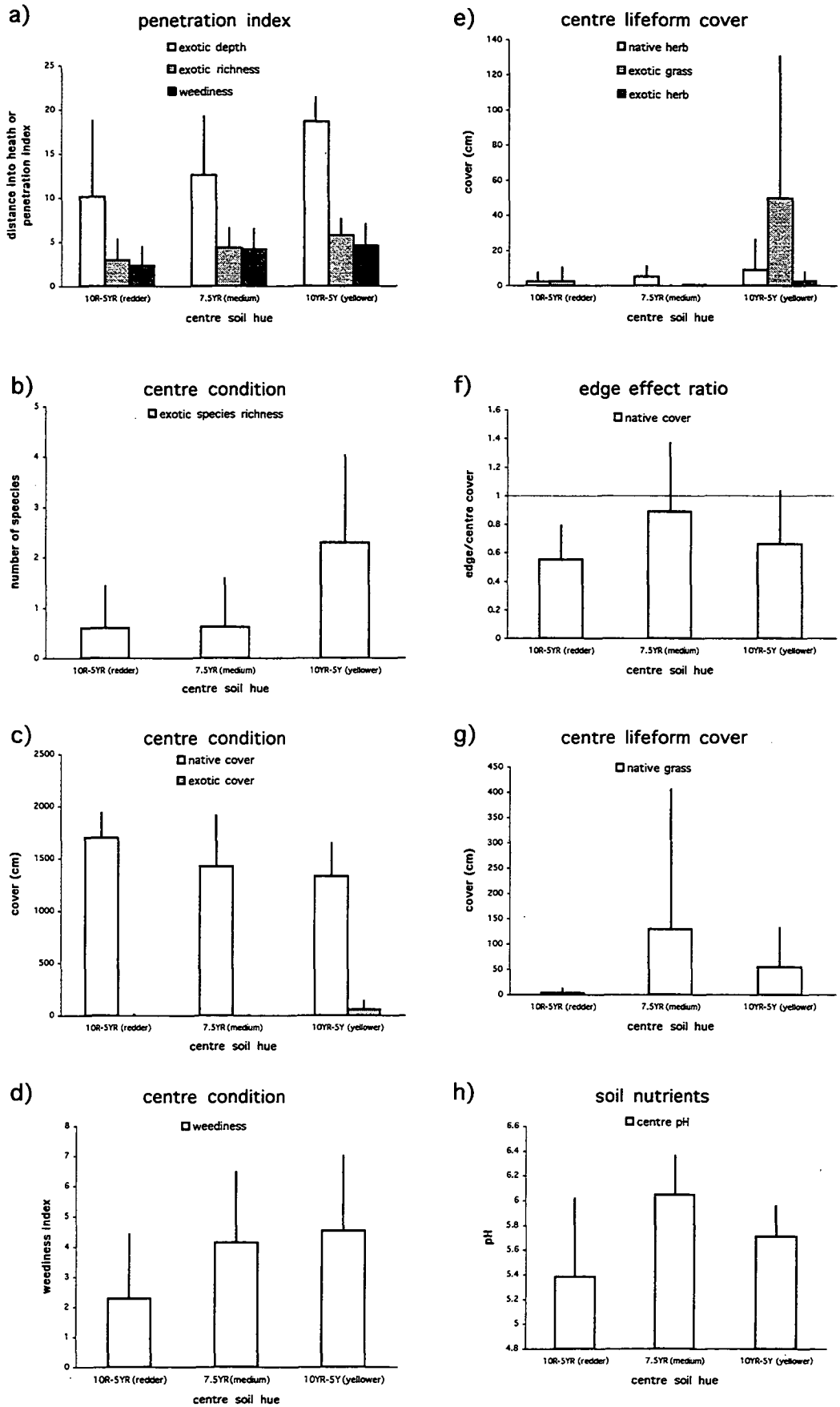


Figure 5.9. Differences in a) penetration indices, b–d) centre condition, e) centre lifeform cover, f) edge effect ratio, g) native grass cover and h) soil nutrients for soil hue in the heathland centre. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness index

the centre (Figure 5.9g). Lower hue at the heath edge was associated with a lower penetration index of exotic species richness, more exotic grasses and less native grasses, and lower pH (Figure 5.10a–c). In the northeast, lower hue in the centre appeared to cause lower native herb % and native grass cover. The same results were found in correlations between continuous rather than class hue data and the condition and lifeform cover variables, at both the edge and the centre.

Summarising these results and substituting pH for hue, it becomes apparent that lower pH, i.e. higher acidity, reduces weed invasion into the heath centre of both exotic herbs and exotic grasses. The more acidic soil in the heath centre allows for more native growth and cover, although not of native grasses and herbs. These hue results support the direct pH findings given above, and show the importance of pH levels in influencing the invasibility of the heathlands in this survey.

Soil value in the heath centre was positively correlated with soil hue and with pH in the heath centre. Soil value at the edge and soil value at the centre were positively correlated with each other. Therefore, any soil value results can be used as another indicator of the soil pH. Value was also negatively correlated with nitrogen levels at both the edge and the centre. At the heath edge, soil value was positively correlated with the edge/centre ratio of native cover and edge native grass % cover. In other words, lighter soils had less native cover (due to less grasses) at the edge compared with the centre, were more acidic and had higher nitrogen levels. In the heath centre, value was positively correlated with exotic cover, exotic cover %, exotic/native cover, bare ground, exotic grass cover and exotic herb cover %, and negatively correlated with native cover and native graminoid cover. Value was also negatively correlated with the edge/centre native richness ratio. So darker soils, with higher alkalinity but lower nitrogen, encouraged exotic cover (both grasses and herbs) and bare ground, but discouraged native cover (particularly graminoids). Soil chroma was discarded as an informative variable due to the lack of variation (there were only two chroma categories) and only two significant but contradictory results.

The final soil variable was texture, which was significantly related to some of the lifeform variables. There were less scats, less native grasses and more exotic grasses in sandier soils at the heath edge, and less native grasses and graminoids in more sandy soils at the heath centre (Figure 5.10d, f). In the Other region, the edge/centre native cover ratio was higher and the edge/centre richness ratio was lower for more sandy soils. Turning these results around, the less sandy (more loamy or clayey) soils seemed to encourage native grass and graminoid growth and also a higher presence of feral and native herbivores. More loamy or clayey soils in the Other region meant a reduction in native cover at the edge compared with the centre and less of a difference between edge and centre native richness. Interestingly, these more loamy or clayey soils did not have higher nutrient levels than the sandy soils. Higher phosphorus levels at the heath edge and

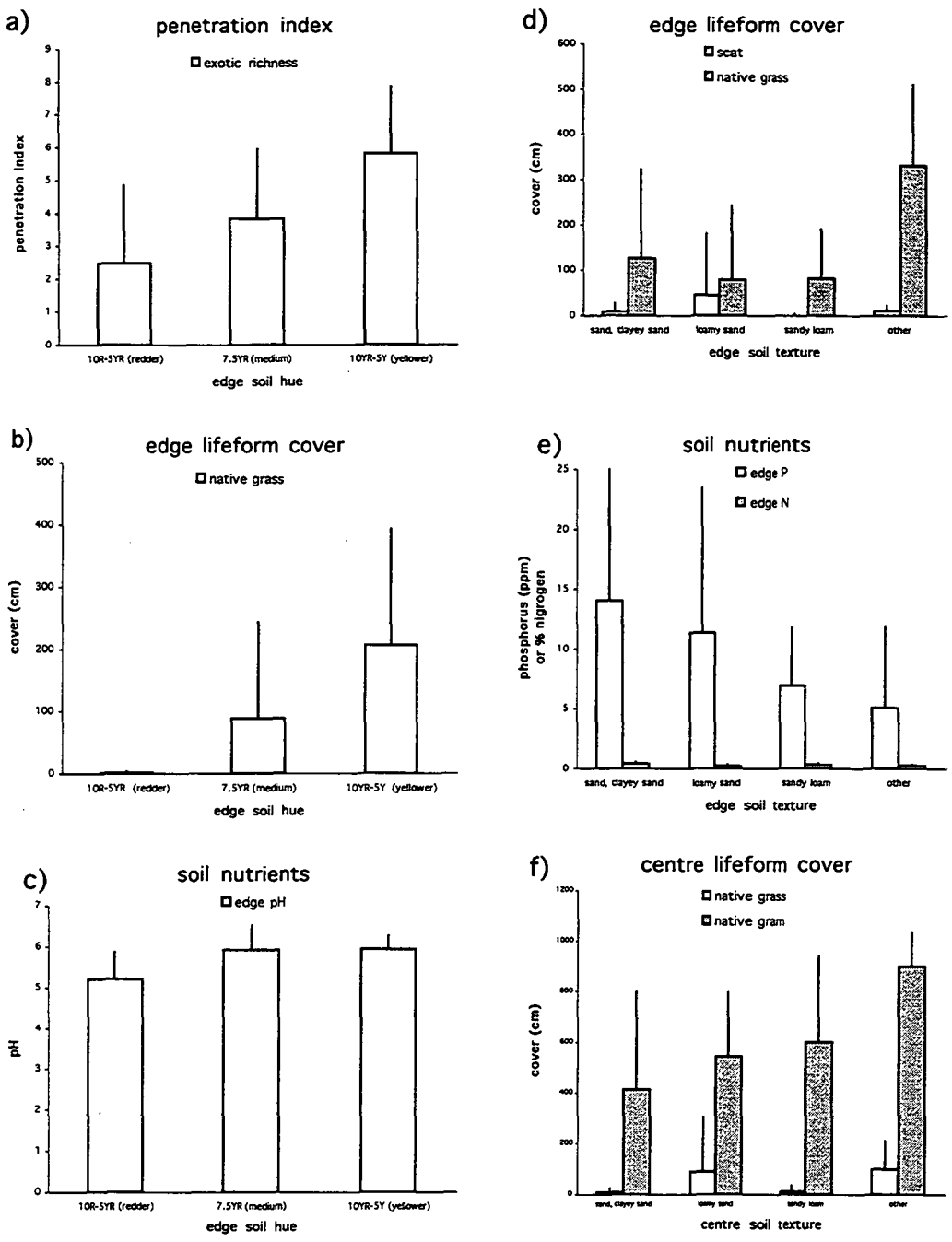


Figure 5.10. Differences in a–c) exotic richness penetration index, edge lifeform cover, and soil nutrients for hue at the heathland edge, d–e) edge and centre lifeform cover and soil nutrients for soil texture at the heath edge, and f) lifeform cover for centre soil texture. 'Gram', graminoid. Error bars show 1 standard deviation

higher nitrogen levels at the heath centre were found with more sandy soils at the edge and centre, respectively (Figure 5.10e). This was also the case for the Northeast region. It is possible that these usually low nutrient soils appear to have higher nutrients due to the increased fertilisation they require.

Pasture management

The land tenure of the pasture influenced various cover variables and the penetration of exotic species into the heath (Table 5.9). The land tenure of the heathland itself (not always the same as for the pasture) also affected exotic and native cover, discussed in the section on heath management, below. In heathland adjacent to freehold or leasehold pasture, weeds did not penetrate as far as in heathland adjacent to pasture in crown land or national parks (Figure 5.11a). This is perhaps opposite of what one might expect, given the supply of exotic propagules and disturbance present in privately owned pasture. Heath next to privately owned land also had a higher edge effect ratio for total cover, meaning a clear boundary between heathland and pasture (Figure 5.11b). The ratio was high not because the edge cover was high but because the centre cover was low (Figure 5.11c), which was caused by lower native grass cover (Figure 5.11d). Native grass cover was also low at the edge in privately owned land (Figure 5.11e) and low in the heath centre for the Other region. Native graminoid cover was high for privately owned land (Figure 5.11d). When land tenure was tested against soil nutrient variables, there was a significantly lower level of phosphorus in the centre of heath adjacent to freehold and leasehold pasture compared with heath adjacent to pasture that was crown land or national park. Phosphorus was also low in the heath centre in the Other region.

The survey covered boundaries with a range of ages, from 7 to 60 years, which were grouped into three classes (Table 5.1). Responses to boundary age were either linear or non-linear. Scat cover increased, native herb cover decreased (Figure 5.11f, g) and exotic grass % decreased with increasing age of the boundary. This implies a relationship where increased grazing pressure results in less native herbs and exotic grasses when boundaries have been established for a long time. Younger boundaries are more likely to have been fertilised and sown more recently, so it makes sense that the amount of exotic grasses at the heath edge was higher. But it is curious that the amount of scats are also lower, when one might expect herbivores to take advantage of the good feed.

Non-linear responses to age include lower exotic herb cover and higher bare ground (Figure 5.11f, g) for medium-aged boundaries (11–20 years). A relationship between pH and age was also found that might explain the vegetation results. Medium-aged boundaries had more acidic soils which would discourage exotic growth (Figure 5.11h). There was also a difference in soil phosphorus in the Northeast region, with levels higher for young and old boundaries and lower for medium-age boundaries. However, there were no significant differences in exotic cover between boundary ages in this region.

There were no differences in the richness, cover or penetration of exotic species between different types of cleared land, however, the amount of native cover and richness did vary. Heath next to pasture had significantly lower native cover in the centre (Figure 5.12a),

Table 5.9. Summary of significant parametric and non-parametric results for general pasture variables

VARIABLES	penetration index exotic depth[1] H DF P	penetration index exotic richness F DF P	penetration index exotic cover F DF P	penetration index exotic weediness F DF P	edge effect ratio total richness F DF P	edge effect ratio native richness F DF P	edge effect ratio exotic richness F DF P	edge effect ratio total cover F DF P
age								
land tenure		**6.98 1 0.011**		**4.20 1 0.046**				**6.66 1 0.013**
pasture type								**3.40 2 0.041**
pasture slashing								

VARIABLES	edge effect ratio native cover F DF P	edge effect ratio exotic cover F DF P	edge condition native richness F DF P	edge condition exotic richness F DF P	edge condition ex:nat richness F DF P	edge condition native cover F DF P	edge condition exotic cover F DF P	edge condition weediness index F DF P
age								
land tenure								
pasture type								
pasture slashing			**10.26 1 0.002**			**4.70 1 0.034**		

VARIABLES	centre condition native richness F DF P	centre condition exotic richness F DF P	centre condition ex:nat richness F DF P	centre condition native cover F DF P	centre condition exotic cover F DF P	centre condition weediness index F DF P	edge lifeform cover scat[1] F DF P	edge lifeform cover bare ground F DF P
age							**9.53 2 0.009**	
land tenure				**6.42 1 0.015**				
pasture type				**5.08 2 0.010**			**11.51 2 0.003**	
pasture slashing								

VARIABLES	edge lifeform cover native fern[1] F DF P	edge lifeform cover native grass[1] F DF P	edge lifeform cover native grass % F DF P	edge lifeform cover native gram F DF P	edge lifeform cover native herb F DF P	edge lifeform cover native herb % F DF P	edge lifeform cover native shrub F DF P	edge lifeform cover exotic grass F DF P
age								
land tenure		**7.42 1 0.006**						
pasture type	**6.77 2 0.034**	**14.31 2 0.001**						
pasture slashing	**8.56 1 0.003**							

VARIABLES	edge lifeform cover exotic grass %[1] F DF P	edge lifeform cover exotic herb F DF P	edge lifeform cover exotic herb %[1] F DF P	centre lifeform cover scat F DF P	centre lifeform cover bare ground F DF P	centre lifeform cover native fern[1] F DF P	centre lifeform cover native grass[1] F DF P	centre lifeform cover native grass % F DF P
age	**6.03 2 0.049**	**6.06 2 0.004**	**7.19 2 0.027**		**4.05 2 0.025**			
land tenure								
pasture type						**6.78 2 0.034**	**10.99 2 0.004**	
pasture slashing								

VARIABLES	centre lifeform cover native gram F DF P	centre lifeform cover native herb[1] F DF P	centre lifeform cover native herb %[1] F DF P	centre lifeform cover native shrub F DF P	centre lifeform cover exotic grass F DF P	centre lifeform cover exotic grass % F DF P	centre lifeform cover exotic herb F DF P	centre lifeform cover exotic herb % F DF P
age		**6.68 2 0.035**						
land tenure	**6.57 1 0.014**							
pasture type								
pasture slashing								

[1] Results for Kruskal-Wallis rank order test

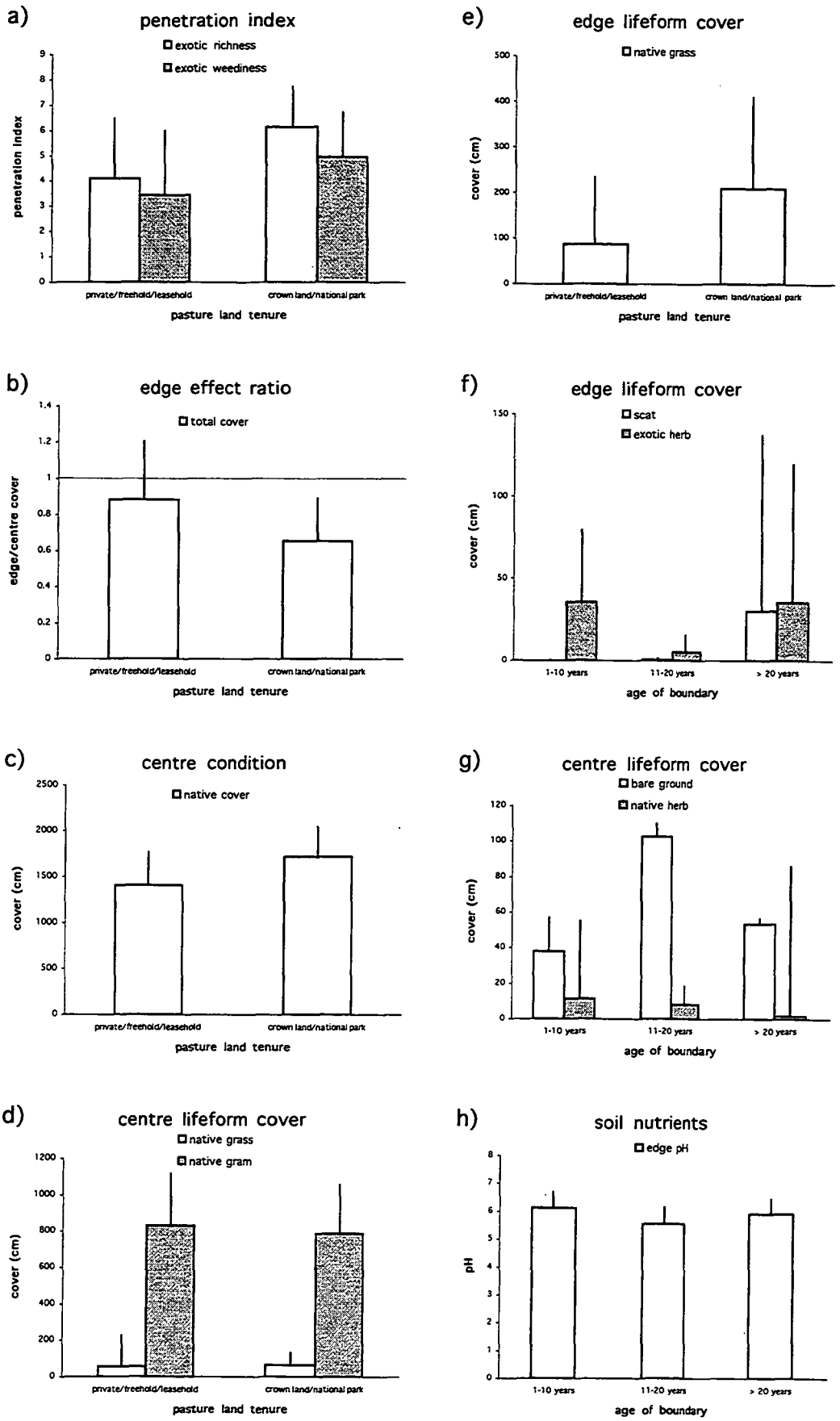


Figure 5.11. Differences in a–e) penetration indices, edge effect ratio, centre condition, and edge and centre lifeform cover for pasture land tenure, and f–h) edge and centre lifeform cover, and soil nutrients for the age of the heath–pasture boundary. 'Gram', graminoid. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness and penetration indices

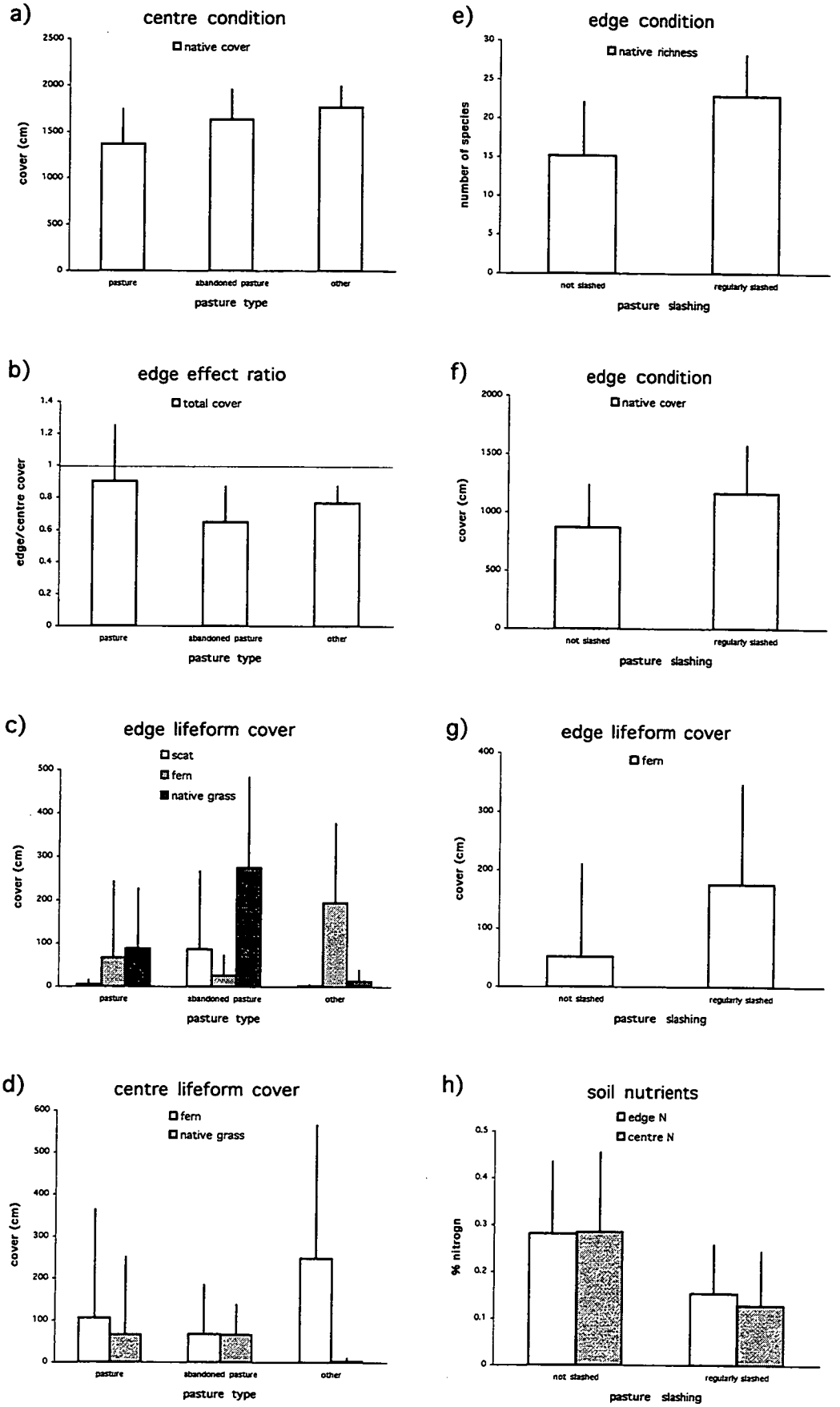


Figure 5.12. Differences in a–d) centre condition, edge effect ratio, and edge and centre lifeform cover for pasture type, and e–h) edge condition, edge lifeform cover and soil nutrients for pasture slashing. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation.

making the edge/centre ratio of total cover about equal (Figure 5.12b), whereas heath next to abandoned pasture had a medium amount of native cover in the heath centre and low edge cover compared with the centre. Neither of the two were significantly different to other types of cleared land (e.g. gardens or ovals). This result shows that the boundaries between pasture and heath were generally much more distinct than between abandoned pasture and heath. Boundaries between other types of cleared land and heathland also appeared to be more diverse in species compared with real pasture. In the Other region, the edge effect ratio for total and native species richness was higher for other types of pasture.

The amount of fern cover in both the edge and the centre was low for both pasture and abandoned pasture but significantly higher for other types of cleared land (Figure 5.12c, d). These other cleared land sites were not found in any particular region (spread between Bruny Island, Flinders Island and Bridport). However, the same fern response for the heath centre was seen in the Northeast region but not in the Other region. The final effect of pasture type was on native grasses and the presence of herbivores, as indicated by scat cover. Native grass cover at both the edge and the centre was highest in abandoned pasture compared with all other types of cleared land, and scat cover at the edge was also highest in abandoned pasture (Figure 5.12c, d). There is no direct causal relationship between these two results but they could still be connected, with herbivore presence higher where there was better feed. The native grass result was the same for the Other region but not for the Northeast.

Regular slashing in the pasture resulted in higher native richness, higher native cover and higher fern cover (Figure 5.12e–g) at the heath edge compared with no slashing. The cover results are likely to be due to *Pteridium esculentum* (Austral Bracken) which would cope with the regular disturbance very well. However, it is unclear why species richness would be higher when regularly slashed, as many heathland species do not favour such physical disturbance. One reason may be that some species were not able to compete in the higher nitrogen environment that was found when the pasture was not slashed (Figure 5.12h).

Grazing

Many of the grazing variables affected the condition, lifeform, invasibility and edge effect variables (Table 5.10), often with associated soil nutrient variations. A medium period of time the pasture was grazed appeared to discourage weed penetration whereas a short or long period did not. This can be seen in the lower depth of exotic penetration, exotic species richness (Figure 5.13a, b) and exotic/native richness for 11–20 years of grazing. The lower presence of exotics could be explained by the more acidic soils that were found for 11–20 years of grazing (Figure 5.13c). Another explanation for this result could be

Table 5.10. Summary of significant results for grazing variables

VARIABLES	penetration index exotic depth[1]			penetration index exotic richness			penetration index exotic cover			penetration index exotic weediness			edge effect ratio total richness			edge effect ratio native richness			edge effect ratio exotic richness			edge effect ratio total cover		
	H	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since grazed																								
no. of years grazed	12.30	3	0.006																					
months in yr grazed	13.41	2	0.001	10.81	2	0.000	6.90	2	0.000	5.12	2	0.011										4.04	2	0.024
cattle																			4.57	1	0.037	5.25	1	0.026
sheep																								
other stock																								
stock class																								
stocking rate																						3.41	3	0.027
wallaby	7.10	2	0.029										5.10	2	0.010	4.42	2	0.017				10.64	2	0.000
wombat	10.03	3	0.018																			3.46	3	0.023
Cape Barren geese																								
native grazing index																								
feral grazing index																						7.07	2	0.002

	edge effect ratio native cover			edge effect ratio exotic cover			edge condition native richness			edge condition exotic richness			edge condition ex:nat richness			edge condition native cover			edge condition exotic cover			edge condition weediness index		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since grazed																								
no. of years grazed										5.11	3	0.004	2.86	3	0.048									
months in yr grazed																								
cattle	3.11	1	0.049																					
sheep																								
other stock				6.65	1	0.013	7.62	1	0.008				4.25	1	0.044				4.83[1]	1	0.028	5.49[1]	1	0.019
stock class							3.94	3	0.014															
stocking rate																								
wallaby	4.89	2	0.012				4.70	2	0.014															
wombat																								
Cape Barren geese																			3.93[1]	1	0.047			
native grazing index																								
feral grazing index																								

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.10. Summary of significant results for grazing variables

	centre condition native richness			centre condition exotic richness			centre condition ex:nat richness			centre condition native cover			centre condition exotic cover[1]			centre condition weediness index[1]			edge lifeform cover scat			edge lifeform cover bare ground[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since grazed										3.09	3	0.039												
no. of years grazed																								
months in yr grazed	4.09	2	0.025	10.04[1]	2	0.007	9.72[1]	2	0.008				11.76	2	0.003	10.96	2	0.004						
cattle				6.99[1]	1	0.008	7.31[1]	1	0.007							8.57	1	0.003						
sheep																								
other stock																								
stock class																								
stocking rate																								
wallaby	4.77	2	0.014							5.21	2	0.010							8.26[1]	2	0.016			
wombat										2.86	3	0.048	8.17	3	0.043							15.46	3	0.001
Cape Barren geese																								
native grazing index																			0.273	+	0.038			
feral grazing index																			9.04[1]	2	0.011			

	edge lifeform cover native fern			edge lifeform cover native grass			edge lifeform cover native grass %			edge lifeform cover native gram			edge lifeform cover native herb[1]			edge lifeform cover native herb %			edge lifeform cover native shrub			edge lifeform cover exotic grass[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since grazed				6.55[1]	2	0.038																		
no. of years grazed	8.06[1]	3	0.045	9.99[1]	3	0.019																		
months in yr grazed																11.54	2	0.003	7.66	2	0.013			
cattle																								
sheep				5.42[1]	1	0.020										4.22	1	0.040						
other stock																7.11	1	0.008	5.54	1	0.039	4.78	1	0.029
stock class																								
stocking rate										3.07	3	0.040												
wallaby				6.06[1]	2	0.048																		
wombat				17.79[1]	3	0.000				3.64	3	0.019												
Cape Barren geese																4.05	1	0.044				4.66	1	0.031
native grazing index	0.414	-	0.001	0.297	+	0.024	0.288	+	0.028															
feral grazing index				8.19[1]	2	0.017																		

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.10. Summary of significant results for grazing variables

	edge lifeform cover exotic grass %[1]			edge lifeform cover exotic herb			edge lifeform cover exotic herb %[1]			centre lifeform cover scat			centre lifeform cover bare ground			centre lifeform cover native fern[1]			centre lifeform cover native grass[1]			centre lifeform cover native grass %[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since grazed													5.08	2	0.011				10.33	2	0.006	3.23	2	0.013
no. of years grazed																10.40	3	0.015	10.27	3	0.016	2.87	3	0.028
months in yr grazed							7.20	2	0.027				4.43	2	0.019									
cattle																			4.64	1	0.031			
sheep																								
other stock	5.42	1	0.020																					
stock class																								
stocking rate																								
wallaby													3.66	2	0.034				7.48	2	0.024	3.23	2	0.038
wombat													3.38	3	0.027				18.15	3	0.000	2.82	3	0.000
Cape Barren geese							7.54	1	0.006				8.00	1	0.007									
native grazing index																								
feral grazing index																			15.22	2	0.000	3.21	2	0.000

	centre lifeform cover native gram			centre lifeform cover native herb[1]			centre lifeform cover native herb %			centre lifeform cover native shrub			centre lifeform cover exotic grass[1]			centre lifeform cover exotic grass %[1]			centre lifeform cover exotic herb[1]			centre lifeform cover exotic herb %[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since grazed	3.36	2	0.044																6.86	2	0.032	6.77	2	0.034
no. of years grazed																								
months in yr grazed													9.90	2	0.007	10.10	2	0.006						
cattle	4.73	1	0.035																					
sheep																								
other stock																								
stock class																								
stocking rate																								
wallaby	8.34[1]	2	0.015																					
wombat																			8.29	3	0.040	9.29	3	0.026
Cape Barren geese				4.14	1	0.042																		
native grazing index																								
feral grazing index																								

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

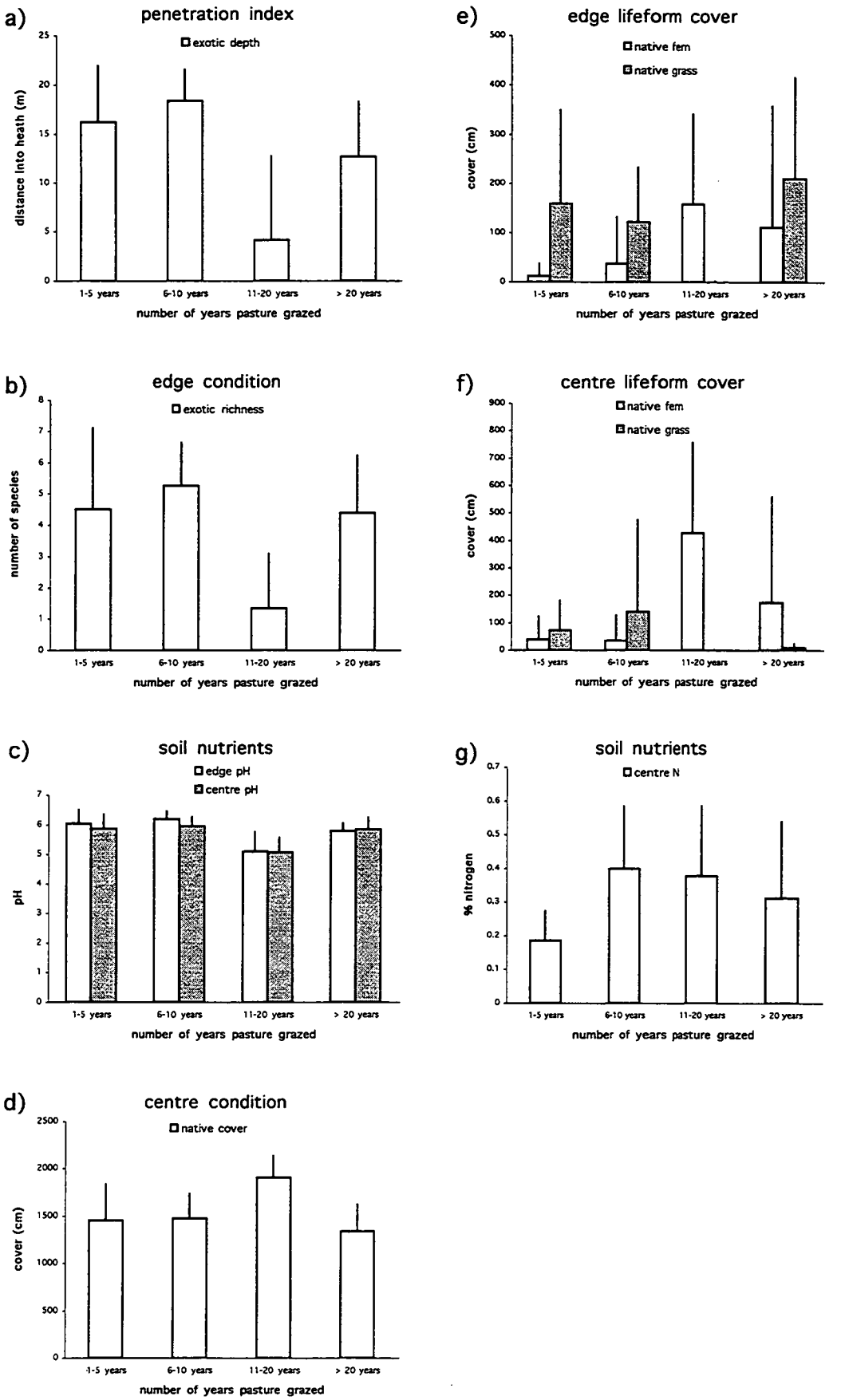


Figure 5.13. Differences in a) penetration index, b) edge condition, c) soil nutrients, d) centre condition, e) edge lifeform cover, f) centre lifeform cover and g) soil nutrients for the number of years the pasture was grazed. Error bars show 1 standard deviation. See text for description of penetration index

regional influences, as most of the sites with 11–20 years of grazing were found in the Other region, and exotic richness varied significantly with region. There were no significant differences in exotics when the two regions were tested separately. Native cover showed the opposite pattern, generally being highest for 11–20 years of pasture grazing. Native cover in the centre, fern cover at the edge, and fern cover at the centre (Figure 5.13d–f) all showed this pattern. Native grasses acted more like the exotics, the cover value being low or absent for 11–20 years of grazing (Figure 5.13e, f). Similar explanations could be used as for the exotic responses just described, that is, either an effect of the soil acidity or regional variation. However, for native grasses, the pattern of variation is more likely to be due to the soil pH than the region, because the same pattern was seen in the Other region when it was tested separately. Nitrogen in the heath centre also varied in a way that was not reflected in richness or cover variables (Figure 5.13g). It is important to note that the time period a pasture has been grazed does not imply the same time period since or between fertilisations, as some sites were fertilised regularly and others were not, regardless of when the pasture was first established for grazing.

For the variable 'number of years since the pasture was grazed', native cover generally increased as the time since grazing increased. This was the pattern for native grass cover at the edge (Figure 5.14a), and native grass, native grass % and native graminoid cover at the centre (Figure 5.14b). Recent pasture grazing can also imply recent heathland grazing, as some pasture owners or managers allowed their stock into the heath itself to graze (discussed in more detail below). These results show that native grasses (less so native graminoids) could be palatable to these stock, resulting in their reduced cover. The same pattern was true for the regions. Fern cover in the centre in the Northeast region and native grass in the centre for the Other region were low for current grazing and high for 6 or more years since grazed. The results for the regions could be due to soil nutrient variations. In the Northeast region, phosphorus and nitrogen were low when fern cover was low and, in the Other region, grass cover was low when the soils were more acidic. The cover of exotic herbs in the heath centre was highest when the pasture had been grazed 1–5 years ago and lowest when the pasture was currently grazed (Figure 5.14c). The low result for current grazing coincided with the highest level of phosphorus at the heath edge (Figure 5.14d).

Exotic richness and cover responded to intermediate levels of grazing, in terms of months in the year. The depth of penetration, richness, cover and weediness penetration indices, centre exotic richness, exotic/native richness, exotic cover, weediness index and exotic grass cover (Figure 5.15a–f) were all lowest for 6–11 months grazed. This pattern is unlikely to be a regional one because none of these variables showed significant variation with region. A simpler way to look at the same results is that weed invasion was generally higher when grazing in the pasture occurred for less than 6 months of the year, as was the total cover edge effect ratio (Figure 5.15g). The same result for the edge effect ratio was seen in the

Northeast region. Part of the reason why cover at the edge was higher relative to the centre could be because bare ground in the centre was higher (Figure 5.15f). It is possible that soil nutrients played a part in the weed invasion: nitrogen levels were higher for less than 6 months of grazing when exotics were also higher (Figure 5.15h). However, this difference was at the heath edge not in the centre, so it may have only indirectly allowed for the build up and penetration of exotics.

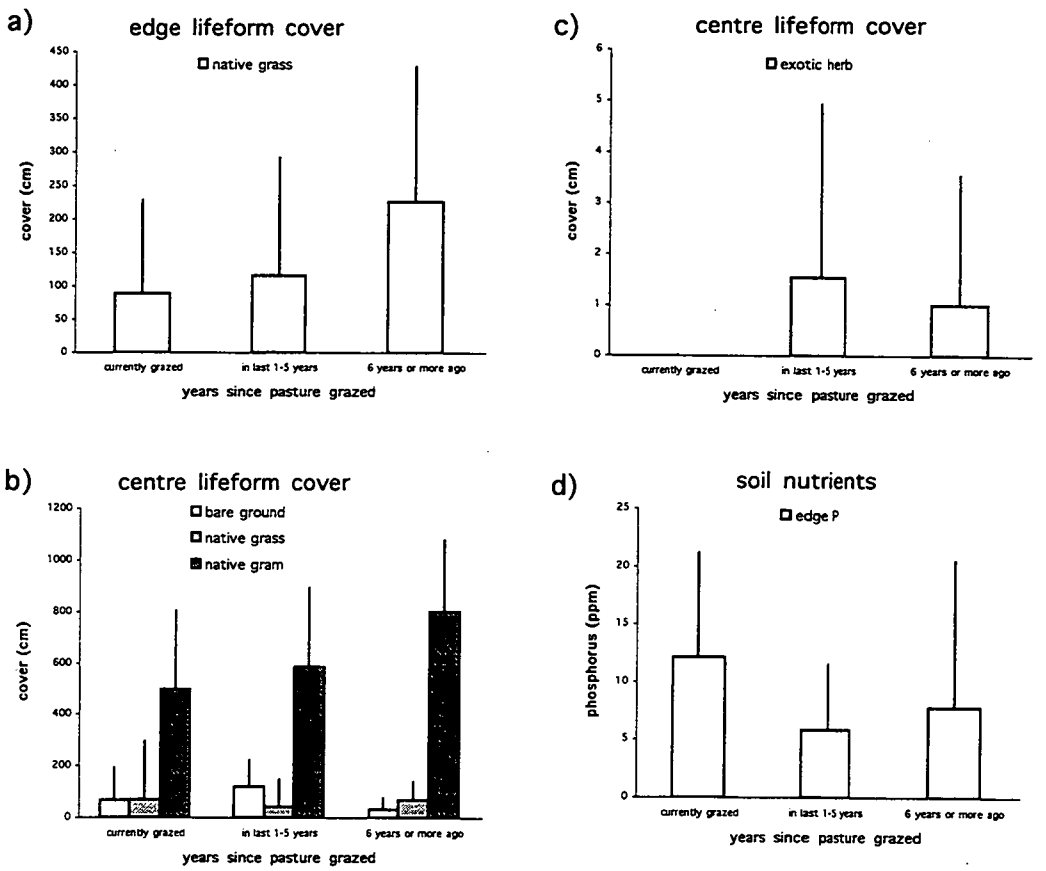


Figure 5.14. Differences in a) edge lifeform cover, b–c) centre lifeform cover and d) soil nutrients for the number of years since the pasture was grazed. 'Gram', graminoid. Error bars show 1 standard deviation.

Native shrub cover at the edge responded in the opposite way to the exotics, being higher for 6–11 months of the year the pasture was grazed (Figure 5.15f). It could be that there is an intermediate amount of grazing disturbance that is beneficial for the natives, allowing a more solid shrub canopy which may then have made it difficult for exotic plants to grow. Other responses from natives occurred in the regions. In the Northeast region, bare ground and native richness were low for more than 6 months grazing; in the East Coast and Waterhouse areas, native richness at the edge was low if grazing was all year; and in the Other region, native grass cover was high for grazing all year. Stock grazing thus leads to more browsing and trampling at the heath edge, which was detrimental to native richness

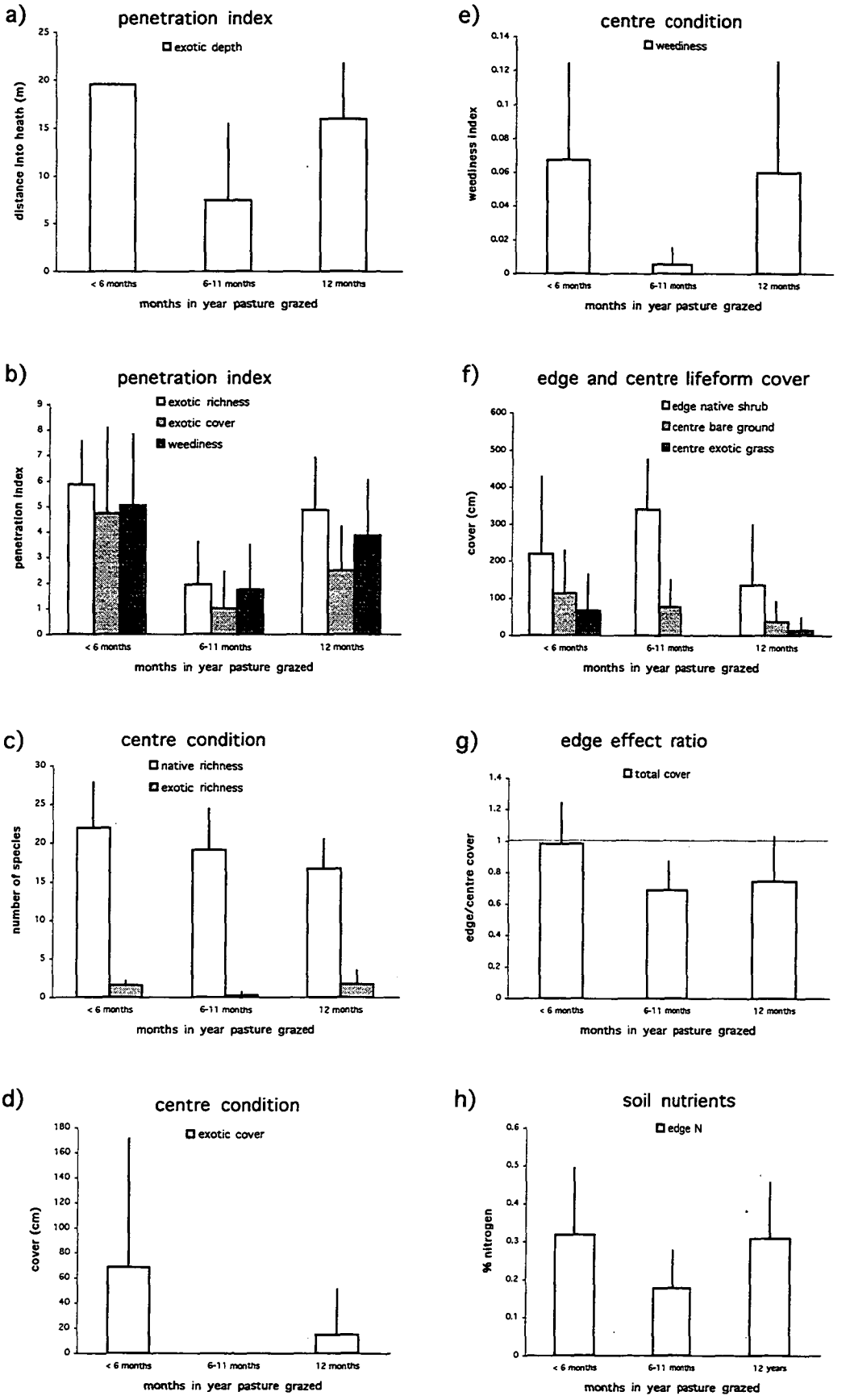


Figure 5.15. Differences in a–b) penetration indices, c–e) centre condition, f–g) edge and centre lifeform cover and h) soil nutrients for the number of months in the year the pasture was grazed. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness and penetration indices

but beneficial for native grasses. A corresponding increase in nitrogen at the edge for the Northeast region provides another type of disturbance.

Weed presence was not affected by the stocking rate in the pasture (measured in dse/ha) but plant cover and soil nitrogen showed significant responses. The total cover edge effect ratio was lowest for 2–5 dse/ha but, for native graminoids and nitrogen, the lowest values were for ≤ 1 dse/ha (Figure 5.16a–c). In the Northeast region, exotic richness at the edge was also lowest for the lowest stocking rate but did not increase linearly, being highest for 6–20 dse/ha. Fern cover in the centre was highest for the lowest stocking rate. Because of the small number of sites for which there were data and the relatively large number of categories, the four stocking rates were combined into two (< 5 and > 5 dse/ha). The significant responses were reduced to only one, the ratio of total cover between edge and centre, which was low for the lower stocking rate and high for the higher stocking rate. In other words, the more stock, the more distinct a boundary between heathland and pasture, perhaps due to a better maintained pasture and more intense grazing management.

Different types of domestic stock affected native species richness (Figure 5.16d). Cattle and sheep alone seemed to have the worst effect on richness at the heath edge, whereas pigs, horses and camels appeared to be relatively harmless. This might be a reflection of the grazing preferences and/or the amount of fencing provided with different stock. The lower nitrogen levels at the heath edge could also have influenced the species richness, allowing more low-nutrient adapted natives to successfully compete (Figure 5.16e). Higher nitrogen levels were found in conjunction with cattle grazing alone (Figure 5.16e, f).

The presence of cattle was associated with only twice as many exotic species at the edge compared with the centre, whereas when they were absent there were four times as many exotic species at the edge compared with the centre (Figure 5.17a). This was due to an increase in exotic richness in the heath centre when cattle were present — rather than a decrease at the edge — specifically the exotic species richness, exotic/native richness and weediness variables (Figure 5.17b, c). Total cover was also higher in the centre relative to the edge when cattle were present, in particular, native grass and graminoid cover were higher (Figure 5.17d, e). There was no significant difference in soil nutrient variables in the heath centre between presence and absence of cattle, although nitrogen was higher at the heath edge when cattle were present (Figure 5.17f). The addition of faeces by the cattle could explain the higher nutrient level. The only variable that differed according to region was native grass cover, and it was also high in the Other region when cattle were present. The ratio of edge to centre total richness was higher, phosphorus and nitrogen were higher and soil pH was more acidic with the presence of cattle in the Other region.

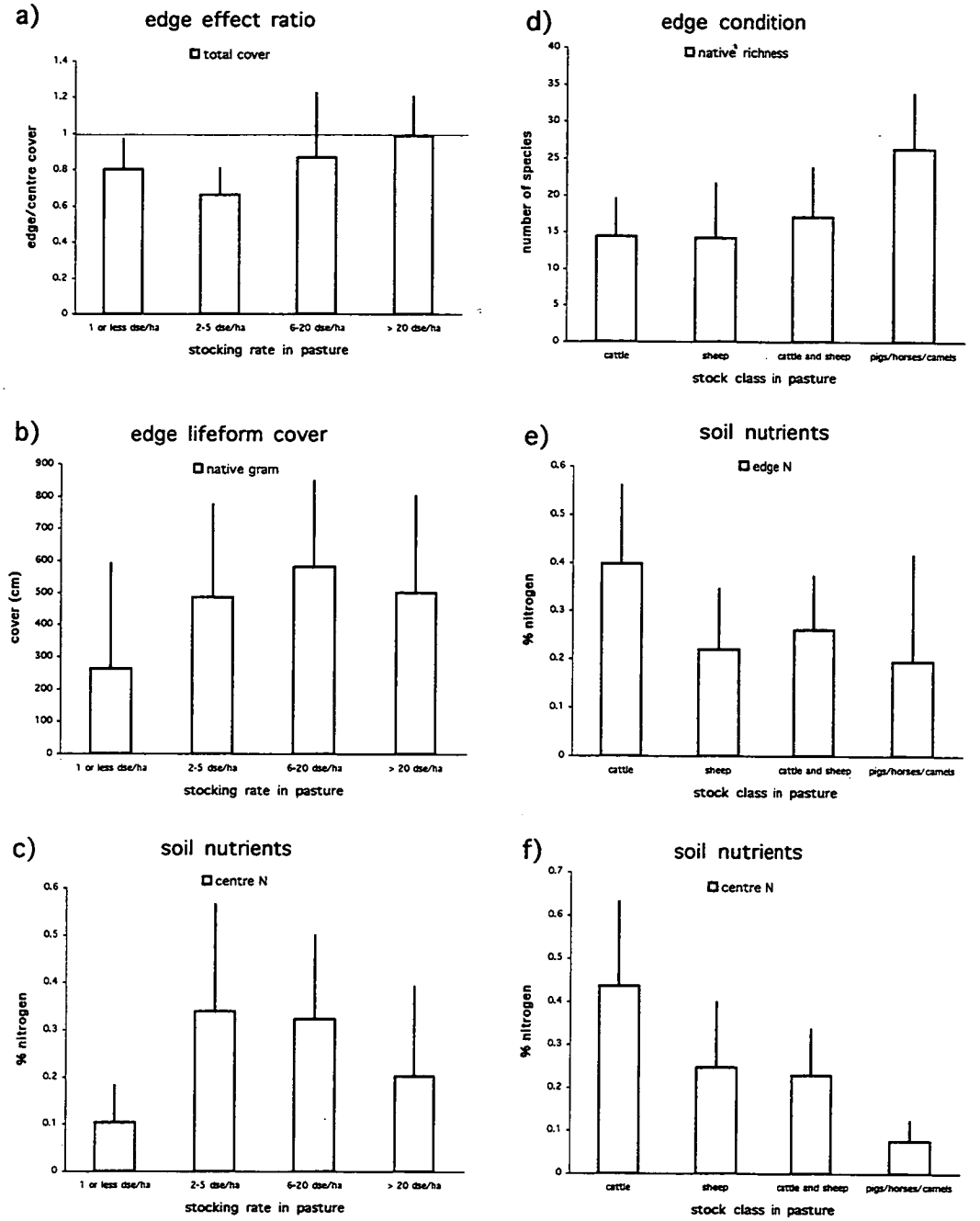


Figure 5.16. Differences in a–c) edge effect ratio, edge lifeform cover and soil nutrients for the stocking rate in the pasture, and d–f) edge condition and soil nutrients for the stock class in the pasture. 'Gram', graminoid. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation.

When sheep were present, edge native grass cover was high (Figure 5.17g) but edge native herb % cover was low. The native herb % result can be ignored because it showed regional variation and, when the regions were tested separately, it responded in the opposite way in the Other region. There was also more native cover at the edge than the centre and lower nitrogen levels in the centre. So the presence of sheep was associated with more native grass cover at the edge and less native herbs and lower nitrogen levels in the heath centre.

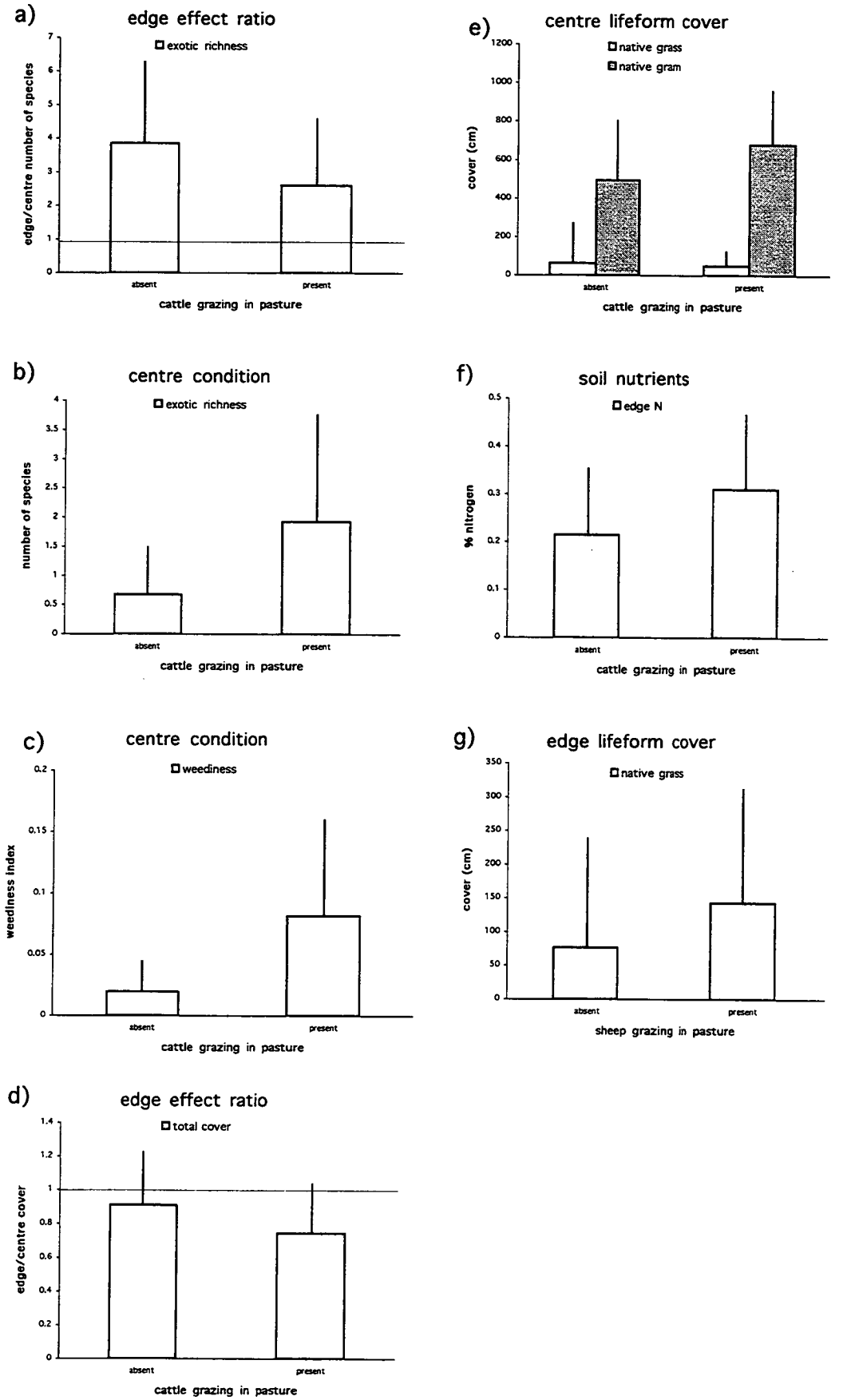


Figure 5.17. Differences in a–f) edge effect ratios, centre condition, edge and centre lifeform cover and soil nutrients for presence or absence of cattle grazing in the pasture, and g) edge lifeform cover for presence or absence of sheep grazing in the pasture. 'Gram', graminoid. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness index

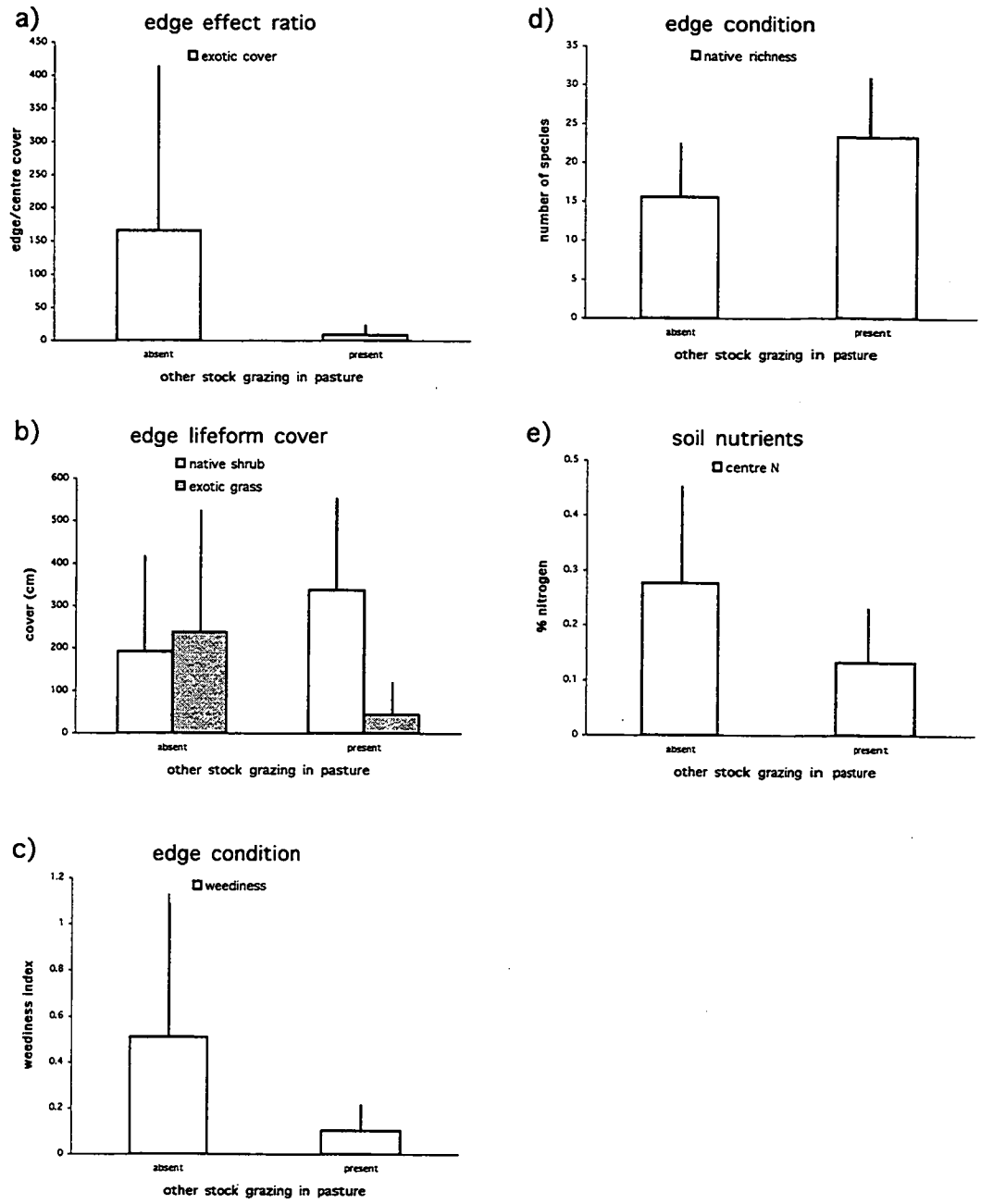


Figure 5.18. Differences in a) edge effect ratio, b) edge lifeform cover, c–d) edge condition and e) soil nutrients for presence or absence of other stock grazing in the pasture. Error bars show 1 standard deviation. See text for description of weediness index

Pigs, camels and horses formed the category 'other stock' and their presence was associated with a much lower exotic cover edge effect ratio than their absence (Figure 5.18a). The difference can be attributed to less exotic cover at the edge, rather than more cover in the centre. Exotic/native richness, exotic cover, weediness, exotic grass cover and exotic grass cover % were all low when other stock were present (Figure 5.18b, c). Native vegetation showed the opposite pattern, with native richness, native shrub cover and native herb % cover at the edge being high when other stock were present (Figure 5.18b, d). Nitrogen levels were also low when other stock were present (Figure 5.18e), but this was found in the

heath centre and not the edge, so it does not explain the lower exotic presence at the edge. Either the other stock preferentially graze on exotic grasses over native shrubs and herbs, or the differences in cover and richness can be attributed to region. The second case is more likely, because most of the variables that responded to the presence of other stock were subject to regional variation, and only one site with other stock present was found in the Northeast, all others were found in the Other region.

Wallaby and wombat grazing had a positive relationship with the depth of penetration of exotics. Weed penetration was not as deep when wallaby grazing was light, whereas moderate or heavy grazing pressure was associated with deeper weed penetration (Figure 5.19a). All other variables that responded to wallaby grazing were native rather than exotic. Lower total and native richness edge effect ratios, edge native richness, centre native cover and centre native graminoid cover were associated with moderate grazing (Figure 5.19b–e). The lower native richness edge effect ratio was due to both lower richness at the edge and higher richness in the centre (Figure 5.19c). The lower edge richness coincided with higher edge phosphorus levels (Figure 5.19f). The native cover edge effect ratio was higher for moderate grazing, due to lower cover in the heath centre and higher bare ground (Figure 5.19g, d, e). The picture is quite complex, but a few possible relationship can be drawn out. Firstly, there may be a relationship between higher edge phosphorus, greater weed penetration and more weed cover at the edge. There may also be a relationship between higher phosphorus and lower native richness at the edge. It is interesting that these relationships occurred for moderate wallaby grazing rather than heavy or severe grazing. Two variables increased as grazing pressure increased: scat cover and native grass cover in the edge and centre (Figure 5.19e, h). Native grass cover in the centre is subject to regional effects but it did not vary significantly with grazing pressure when the regions were tested separately. The variables that were significantly different in the regions were higher phosphorus levels and relatively higher total richness in the heath centre (compared with the edge) with higher levels of grazing, and high exotic richness and exotic/native richness ratios for moderate grazing in the Other region. Moderate grazing pressure does seem to have the most detrimental effect on the heathland in terms of weed invasion.

Wombat grazing showed some similar patterns to wallaby grazing: weed penetration was deepest with moderate grazing (Figure 5.20a). This result was very obvious in the field, where weeds were present along the bare ground of most native mammal trails under the shrubs but not to either side (see Native Mammal Trail section below). Again, moderate grazing had the greatest effect, with the highest ratio of edge/centre total cover (due to low centre native cover), and highest cover of exotics, specifically exotic herbs (Figure 5.20b–d). In the Northeast region, exotic/native richness at the edge was highest for moderate grazing. For the dataset as a whole, these responses could have been due to

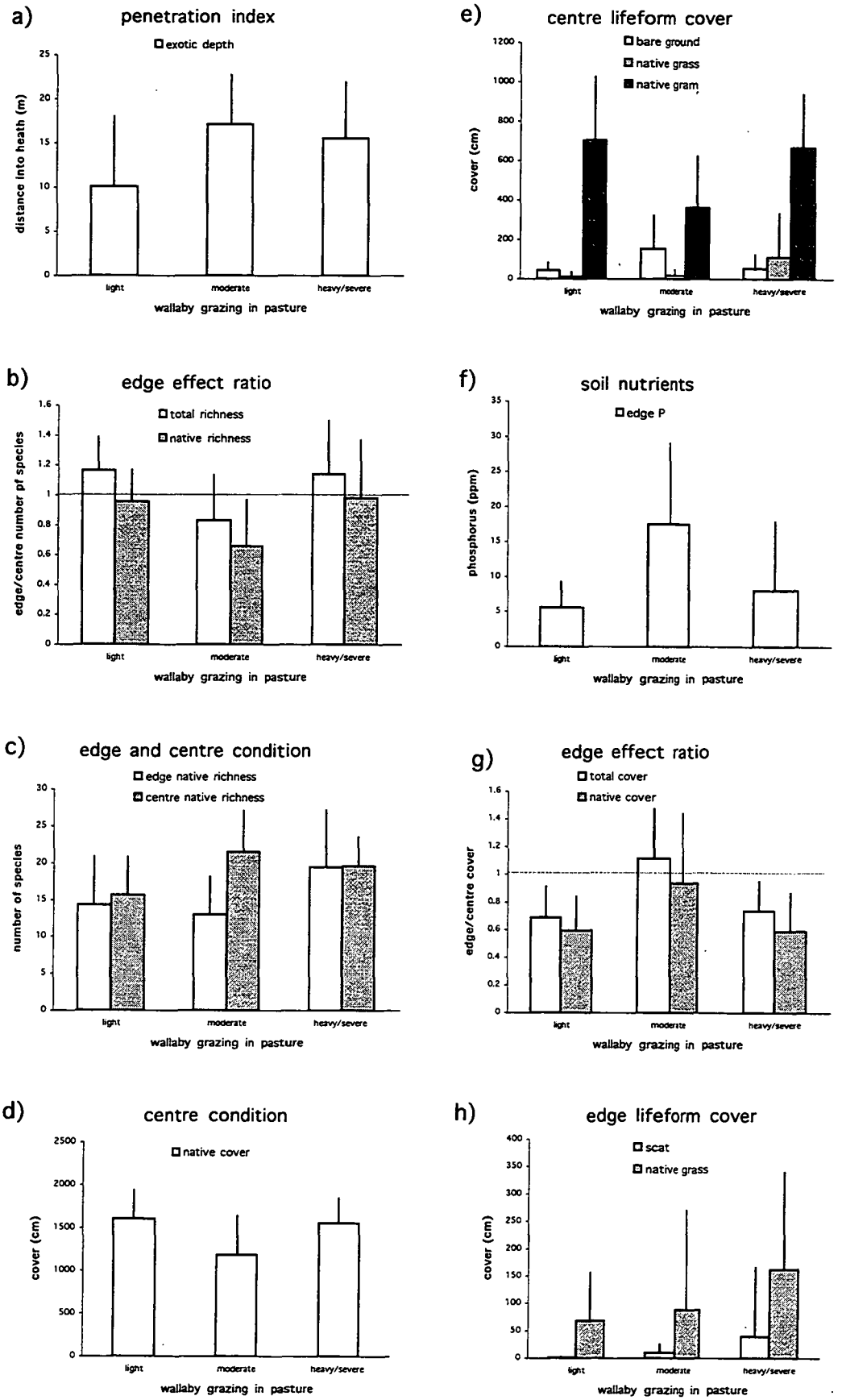


Figure 5.19. Differences in a) penetration index, b) richness edge effect ratios, c–d) edge and centre condition, e) centre lifeform cover, f) soil nutrients, g) cover edge effect ratios and h) edge lifeform cover for the level of wallaby grazing in the pasture. 'Gram', graminoid. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of penetration index

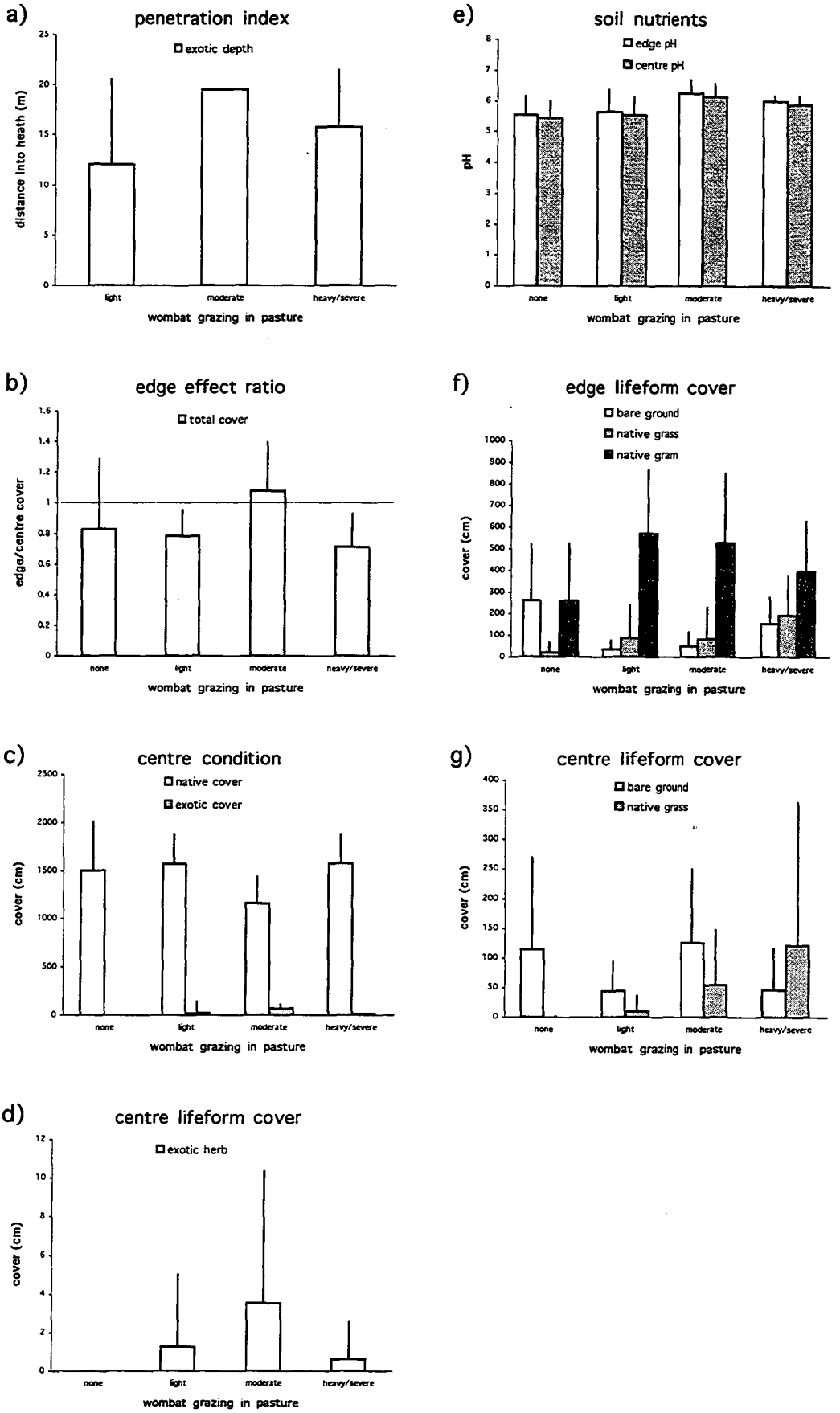


Figure 5.20. Differences in a) penetration index, b) edge effect ratio, c–d) centre condition and lifeform cover, e) soil nutrients, and f–g) edge and centre lifeform cover for the level of wombat grazing in the pasture. 'Gram', graminoid. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation. See text for description of penetration index

changes in soil variables, as both edge and centre pH were also highest for moderate grazing (Figure 5.20e). A relationship may exist between high exotic cover, low native cover and more alkaline soils. Why this occurs for moderate wombat grazing implies another complex relationship between native grazing and weed invasion. High levels of wallaby and wombat grazing are likely to occur where there is good pasture, which means domestic stock may also be present. Perhaps a very high presence of wallaby, wombat and stock preferentially graze the exotic species, resulting in lower levels of weed invasion.

Native grasses at the edge and centre increased as the wombat grazing pressure increased, the same response as seen with wallaby grazing (Figure 5.20f, g). Centre native grass cover is subject to regional influences, but testing the regions separately showed that the same pattern held for the Other region. It is speculated that there is again a relationship between better pasture and more native mammal grazing, and that better pasture may be due to slightly more fertile soils resulting in more native grass cover in the heathland. More alkaline soils with higher nitrogen levels coincided with increased wombat grazing in the Other region, adding support for this speculation. Native graminoids and bare ground at the edge could be responding to a different set of factors, as graminoids were high for mid levels of grazing and bare ground was low (Figure 5.20f). Bare ground in the centre was rather erratic, perhaps reflecting the presence of native mammal trails through the heath (Figure 5.20g).

The response to the presence of Cape Barren Geese grazing was simple. Exotic cover, exotic grass cover and exotic herb % cover at the edge were high when geese were present (Figure 5.21a). Native herb cover at the edge and centre and bare ground at the centre were all low when geese were present (Figure 5.21b). In the East coast and Waterhouse areas, and in all areas excluding Flinders, native richness at the edge was also low when geese were present. The presence of geese was also associated with high phosphorus and nitrogen levels (Figure 5.21c, d).

The native grazing index as a continuous variable was not significant against the ratios or the penetration indices but there was a positive correlation between the grazing index and scat cover, native grass cover and native grass % at the heath edge (Table 5.10). This shows that counting scats was a reasonably good way to assess the grazing pressure. There was a negative correlation between the grazing index and native fern cover at the edge, possibly due to *Pteridium esculentum* (Austral Bracken) being lower in sites that were better for grazing.

The feral grazing index showed intermediate grazing effects, except that grazing by rabbits and hares was never heavy or severe. When grazing was light there was the same amount of cover between the edge and centre, and for none or moderate grazing the ratio was lower and thus the boundary was more diffuse (Figure 5.21e). As with wallaby and

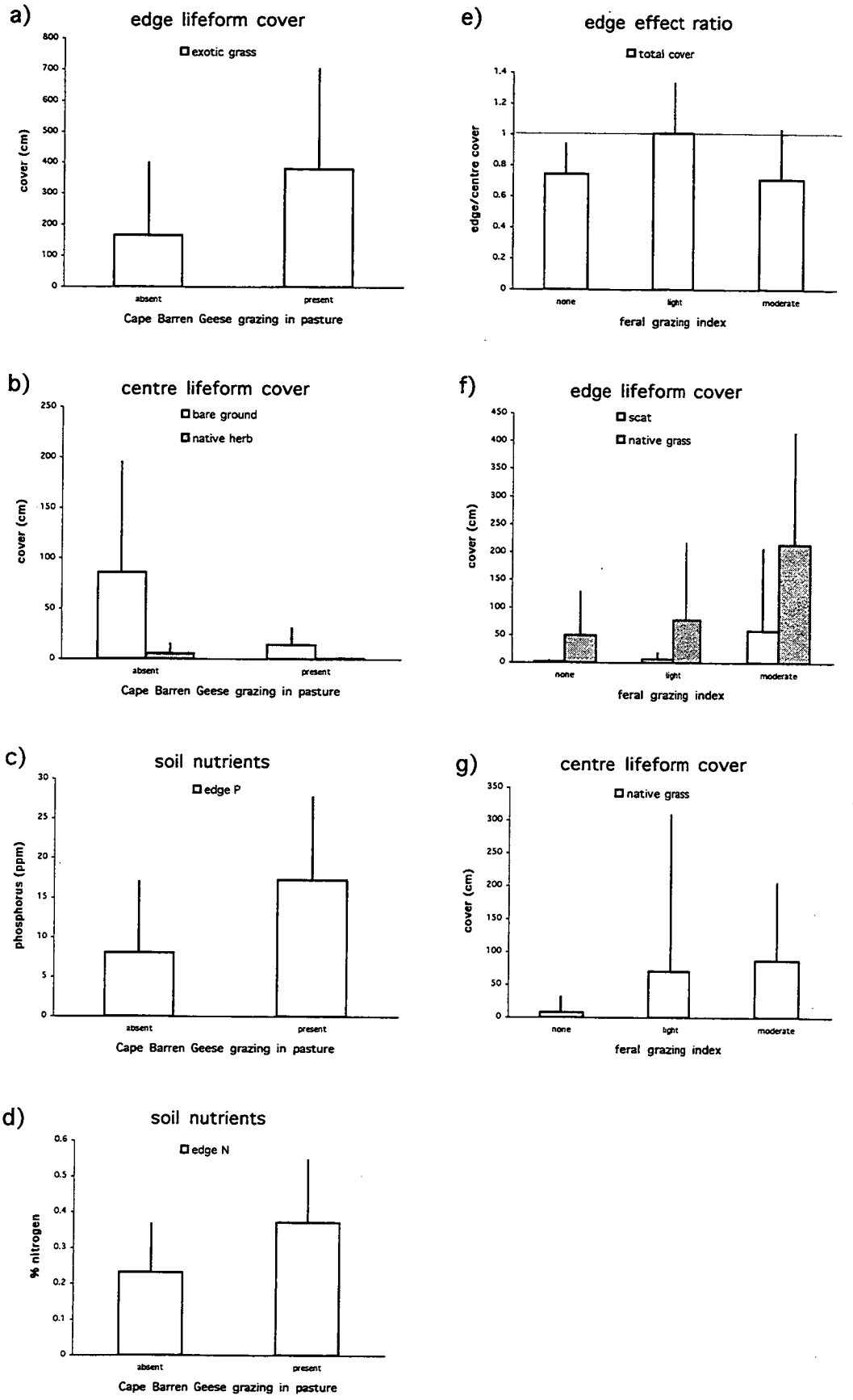


Figure 5.21. Differences in a–d) edge and centre lifeform cover and soil nutrients for presence or absence of Cape Barren Geese grazing in the pasture, and e–g) edge effect ratio, and edge and centre lifeform cover for the level of feral grazing in the pasture. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation.

wombat grazing, scat cover, edge native grass cover, centre native grass cover and centre native grass % cover were all higher with a higher feral grazing index score (Figure 5.21f, g). Centre native grass cover was also higher with higher feral animal grazing in the Other region. There were no differences in soil nutrients that could explain these results. It could be that the disturbance load of feral herbivores is not great enough to cause a major loss of condition of the heath.

Fertilising

The five fertilising variables all had different effects on soil nutrients, weed invasion, edge effects and heath condition (Table 5.11). There was a significant difference in phosphorus levels at the heath edge with years since fertilisation. Phosphorus levels declined as the number of years since the pasture was fertilised increased (Figure 5.22a). This shows that the soil sampling did adequately indicate the addition of fertiliser, and also indicates that phosphorus levels did not remain high in the soil for more than a year. The ratio of edge/centre total cover did not seem to be related to the phosphorus levels, as it was highest for 2–5 years since fertilised (Figure 5.22b). This was probably due to a decrease in native shrub cover in the heath centre, rather than an increase in total cover at the edge cover (Figure 5.22c). Bare ground was highest when native shrub cover was lowest (Figure 5.22c). These responses are unlikely to be due to region, as none of the variables were subject to regional effects, and the different categories were spread evenly over the two regions. The changes in the total cover edge effect ratio were probably related to the intensity of the pasture management, as there was a more diffuse boundary when the years since the pasture was fertilised increased (Figure 5.22b). The regions also showed greater native canopy cover when there was more time since fertilising, with higher centre fern cover in the Northeast and higher centre grass cover in the Other region.

The number of years between fertilisations in the pasture is another measure of the amount of nutrients that might be added to the heathland. However, it must be remembered that the years between fertilisations is different to the years since fertilising — a site with 10 years between fertilising might have been fertilised in the last year. An intermediate number of years (2–10) had the most effect. For this period, the total edge effect richness ratio was lowest, but the total and native cover edge effect ratios, native cover in the centre and bare ground at the edge were all highest (Figure 5.23a–d). There was no similar pattern of soil nutrient differences that might explain these results. Although the different categories were evenly spread over the two regions, the total richness and native cover edge effect ratios are subject to regional differences. Separate testing of the regions showed the edge/centre native cover ratio decreasing as time between fertilisations increased, in other words, the boundary became more gradual. Nitrogen levels at the edge also tended to decrease over time, and this could have caused the canopy cover at the edge

Table 5.11. Summary of significant parmetric and non-parametric results for fertilising variables

VARIABLES	penetration index exotic depth[1] H DF P	penetration index exotic richness F DF P	penetration index exotic cover F DF P	penetration index exotic weediness F DF P	edge effect ratio total richness F DF P	edge effect ratio native richness F DF P	edge effect ratio exotic richness F DF P	edge effect ratio total cover F DF P
years since fertilised								**3.89 3 0.030**
yrs between fertilising					**5.13 2 0.010**			**11.21 2 0.000**
season fertilised								
fertiliser application								
lime		**5.73 1 0.022**						
amount of super								

	edge effect ratio native cover F DF P	edge effect ratio exotic cover F DF P	edge condition native richness F DF P	edge condition exotic richness F DF P	edge condition ex:nat richness F DF P	edge condition native cover F DF P	edge condition exotic cover F DF P	edge condition weediness index F DF P
years since fertilised								
yrs between fertilising	**5.76 2 0.006**							
season fertilised								
fertiliser application								
lime								
amount of super								

	centre condition native richness F DF P	centre condition exotic richness F DF P	centre condition ex:nat richness F DF P	centre condition native cover F DF P	centre condition exotic cover F DF P	centre condition weediness index F DF P	edge lifeform cover scat F DF P	edge lifeform cover bare ground[1] F DF P
years since fertilised								
yrs between fertilising	**3.51 2 0.042**			**3.35 2 0.047**				**7.98 2 0.019**
season fertilised								
fertiliser application								**11.78 1 0.001**
lime								
amount of super								

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.11. Summary of significant parmetric and non-parametric results for fertilising variables

	edge lifeform cover native fern			edge lifeform cover native grass[1]			edge lifeform cover native grass %			edge lifeform cover native gram[1]			edge lifeform cover native herb			edge lifeform cover native herb %[1]			edge lifeform cover native shrub			edge lifeform cover exotic grass		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since fertilised																								
yrs between fertilising																								
season fertilised																								
fertiliser application																6.51	1	0.011						
lime										4.77	1	0.029				3.89	1	0.049						
amount of super				9.45	1	0.002																		

	edge lifeform cover exotic grass %[1]			edge lifeform cover exotic herb			edge lifeform cover exotic herb %			centre lifeform cover scat			centre lifeform cover bare ground			centre lifeform cover native fern[1]			centre lifeform cover native grass			centre lifeform cover native grass %		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since fertilised													4.16	3	0.013									
yrs between fertilising																								
season fertilised																								
fertiliser application																								
lime	5.40	1	0.020													4.86	1	0.027						
amount of super																						0.410	-	0.012

	centre lifeform cover native gram			centre lifeform cover native herb			centre lifeform cover native herb %			centre lifeform cover native shrub[1]			centre lifeform cover exotic grass			centre lifeform cover exotic grass %			centre lifeform cover exotic herb[1]			centre lifeform cover exotic herb %[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since fertilised										9.07	3	0.028												
yrs between fertilising																			8.28	2	0.016	8.57	2	0.014
season fertilised																								
fertiliser application																								
lime																								
amount of super																								

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

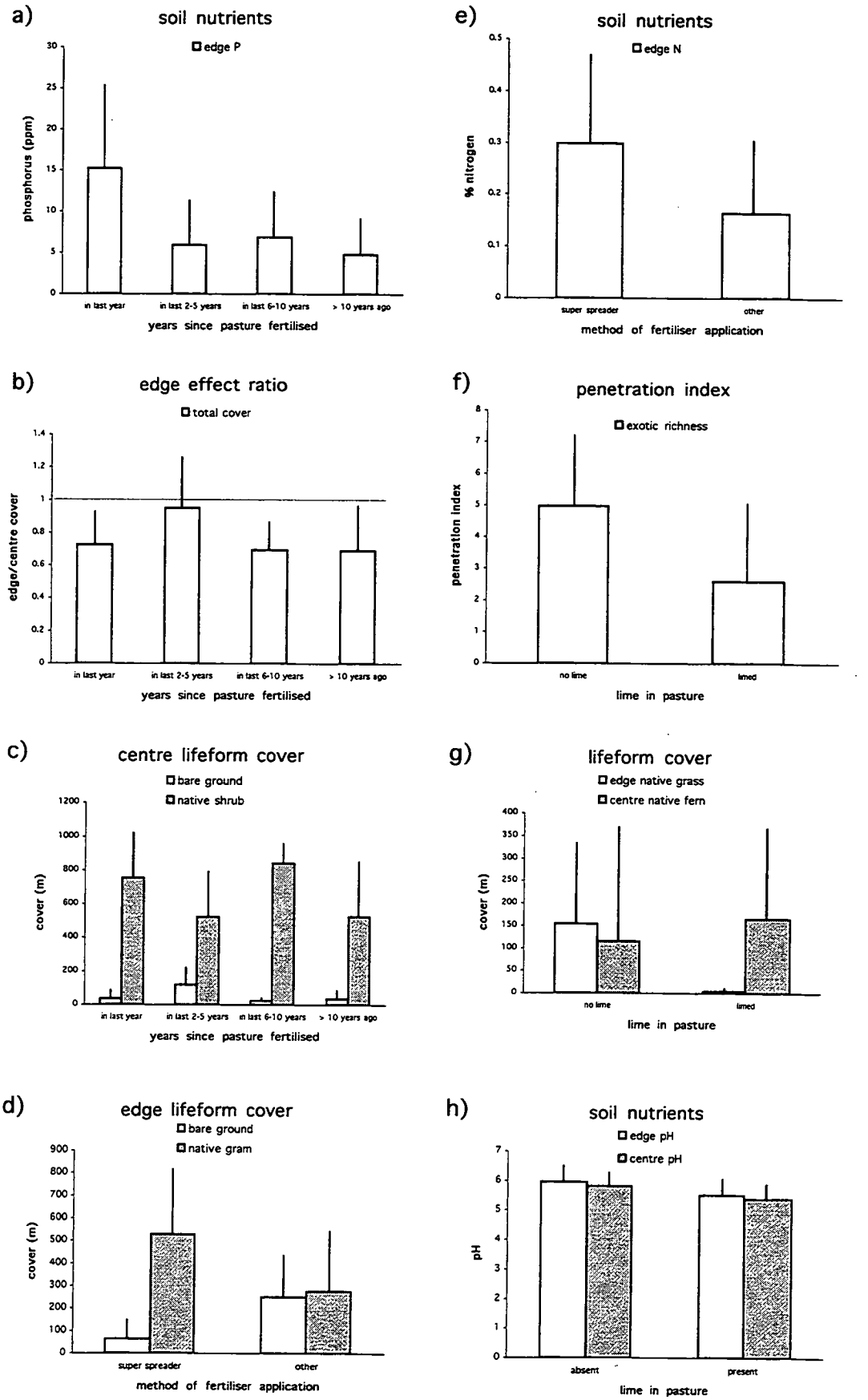


Figure 5.22. Differences in a–c) soil nutrients, total cover edge effect ratio and centre lifeform cover for the number of years since the pasture was fertilised, d–e) edge lifeform cover and soil nutrients for the method of fertiliser application in the pasture, and f–h) penetration index, edge and centre lifeform cover and soil nutrients for whether or not the pasture was limed. 'Gram', graminoid. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation. See text for description of penetration index

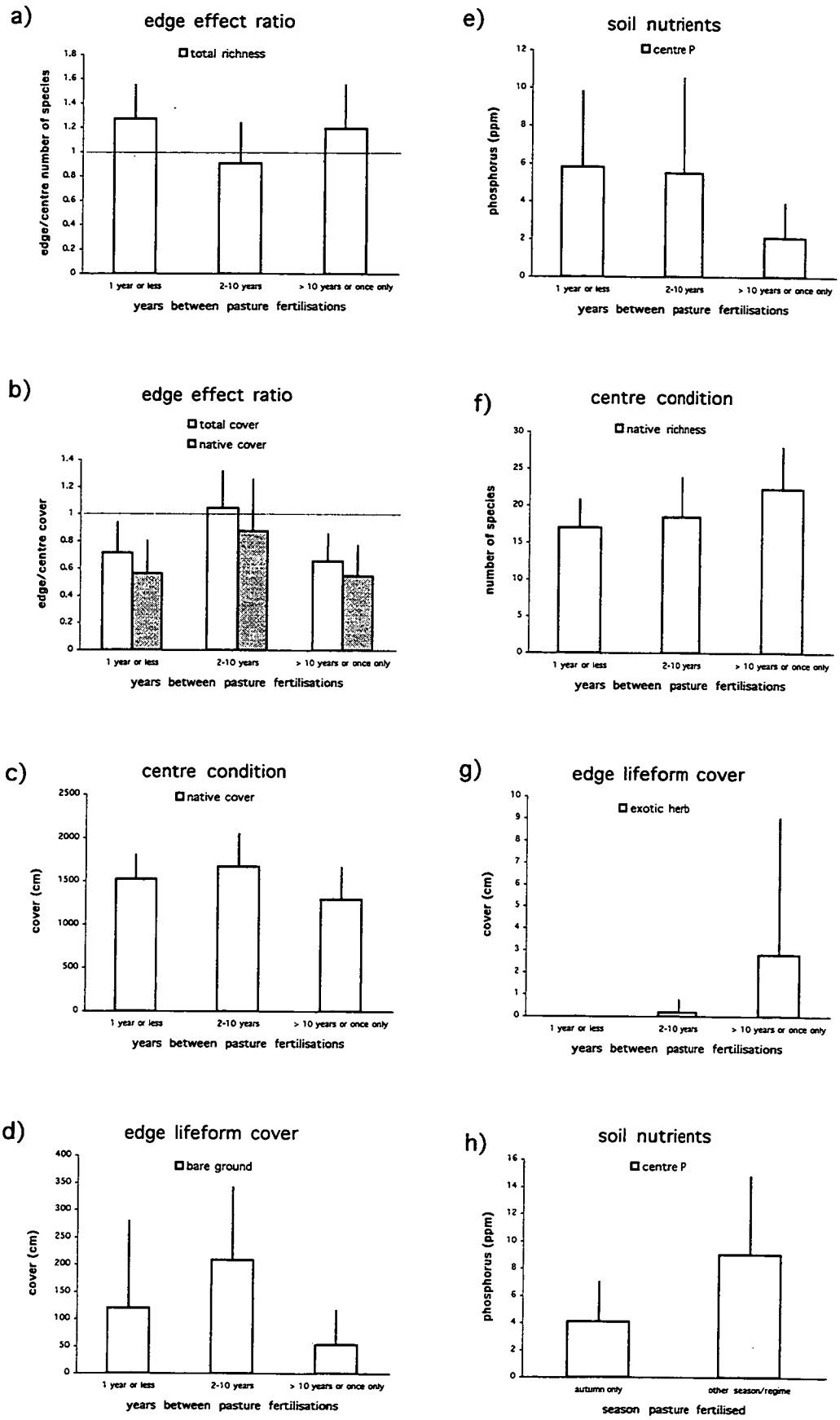


Figure 5.23. Differences in a–b) edge effect ratios, c) centre native cover, d) edge lifeform cover, e) soil nutrients, f) centre native richness, and g) edge lifeform cover for the number of years between fertilisations in the pasture; and h) soil nutrients for the season the pasture was fertilised. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation.

to decline. The other responses to the number of years between fertilisations may be inter-related. Phosphorus in the centre declined as the number of years increased, but native richness increased (Figure 5.23e, f). A longer time between fertilisations means less phosphorus being added to the site, and this might mean native species are better able to compete. However, exotic herb cover at the heath edge also increased as time between fertilisations increased, which is the opposite of what one would expect if more nutrients encouraged weed invasion (Figure 5.23g). Exotic herb cover is also subject to regional effects and it could be this influence that is stronger, or else the time since fertilising is more important.

The season the fertilising took place did not have much effect on the heathland condition. Only two variables responded: phosphorus at the heath centre, which was lower for autumn (Figure 5.23h) and native herb % cover at the edge which was also lower in autumn. For autumn sowing in the Northeast region, exotic cover at the edge was high and the native richness edge effect ratio was roughly even between edge and centre, whereas the ratio was higher for seasons other than autumn. Although there is no causal relationship implied, there could be a connection between the low native richness and high weed cover found at the edge of the heath. Autumn was the most common time of fertilising and only seven sites were fertilised at a different time of year (usually spring). The 'other season' sites were spread evenly over the different regions, so there is unlikely to be a regional effect. It is unclear why herb cover should be lower for autumn fertilising, unless this is related to the timing of the fieldwork. If that was the case then herb cover should have also respond to time since fertilised, but there was no significant response.

The method of applying fertiliser to the pasture mostly affected conditions at the heath edge. Native graminoid and native herb % cover were high and bare ground and exotic richness in the Northeast were low when fertilisers were applied with a super spreader compared with any other method (Figure 5.22d). In all areas excluding Flinders Island, native richness in the centre was low when a super spreader was used. There was no difference in phosphorus levels with method of application, but nitrogen levels at the edge were also high for the super spreader method (Figure 5.22e). This was the case for both the entire dataset and for the Northeast region. As explained in the soil section, nitrogen was rarely applied as a fertiliser but sown as clovers, and these seeds may have been applied in a super spreader along with the fertilisers (see 'method of applying seeds' in Sowing section below). It is possible that some graminoids were able to produce a larger canopy due to the increased nitrogen levels.

The addition of lime to the pasture had a negative relationship with the exotic species richness penetration index (Figure 5.22f), although the sample was small (lime was used in only six sites). The addition of lime was also associated with low edge native grass cover and high centre fern cover (Figure 5.22g). In other words, a more alkaline pasture was

associated with fewer different weed species invading the heath, which is opposite to the soil results usually observed which show that increasing alkalinity in the centre of the heath encouraged weed penetration. Adding lime should mean more alkaline soils in the pasture, however when the pH in the pasture was tested against the addition of lime there was no significant difference. When pH at the heath edge and centre was tested against the addition of lime the pH was more acidic when limed (Figure 5.22h). This could be accounted for if one considers that lime is most likely to be added to sites that were particularly acidic, and perhaps at the six limed sites the liming did not reach the edge of the heath. If this argument is correct, then the lower exotic species penetration index in limed sites is explained by the higher acidity at the edge of the heath. Therefore, native grasses were lower and native fern cover higher when the soils were more acidic. There were not enough lime sites to look at the regions separately.

Sowing

The total cover edge effect ratio and native grass cover responded to the number of years since the pasture was sown (Table 5.12). There did not appear to be any pattern in the way the total cover edge effect varied (Figure 5.24a), except that it was particularly low at the edge when it had been a long time since the pasture was sown. This could be due to the heathland plants having more time to reinvade the pasture, creating a graded boundary. Native grass cover at the edge was highest when it had been more than 20 years since last sown (Figure 5.24b). This was the same for the Other region. Also in the Other region, edge and centre pH became more alkaline as time since sown increased. The native grasses may prefer these less acidic soils or their response could be due to a combination of the frequency of fertilising (done in conjunction with sowing) and the various grazing variables. Native grass cover in the centre acted differently (Figure 5.24b), but was subject to regional effects and thus best discarded. Phosphorus levels in the centre also responded, with the highest level for 6–10 years since sown, and very low levels for the most recent sowing category (Figure 5.24c). It is hard to explain this last result, as usually sowing and fertilising occurred together.

The number of times a pasture was sown was divided into two classes, sown only once since cleared and sown more than once. At the heath edge, bare ground was high if the pasture had been sown only once (Figure 5.24d). This makes sense, as more frequent sowing is likely to result in more exotic cover and thus less bare ground. When the pasture had been sown only once there was a very low ratio of total and native cover at the edge of the heath compared with the centre, but if it had been sown more than once the ratio increased, although not to the point of the edge having an equal cover to the centre (Figure 5.24e). More regular sowing implies a more intensively managed pasture and one might expect that the boundary would be consequently more distinct. Although native cover is subject to

Table 5.12. Summary of significant parametric and non-parametric results for sowing variables

VARIABLES	penetration index exotic depth[1] H	DF	P	penetration index exotic richness F	DF	P	penetration index exotic cover F	DF	P	penetration index exotic weediness F	DF	P	edge effect ratio total richness F	DF	P	edge effect ratio native richness F	DF	P	edge effect ratio exotic richness F	DF	P	edge effect ratio total cover F	DF	P
years since sown																						**10.05**	**3**	**0.018[1]**
no. of times sown																						**7.06**	**1**	**0.011**
season sown	**7.62**	**1**	**0.006**																					
cocksfoot													**4.78**	**1**	**0.036**									
fescue																			**7.77**	**1**	**0.009**			
subclovers																						**6.13**	**1**	**0.018**
no. of different seed																**3.10**	**3**	**0.042**	**5.15**	**3**	**0.005**			
seed application																								

VARIABLES	edge effect ratio native cover F	DF	P	edge effect ratio exotic cover F	DF	P	edge condition native richness F	DF	P	edge condition exotic richness F	DF	P	edge condition ex:nat richness F	DF	P	edge condition native cover F	DF	P	edge condition exotic cover F	DF	P	edge condition weediness index F	DF	P
years since sown							**6.50**	**1**	**0.019**															
no. of times sown	**4.09**	**1**	**0.049**																					
season sown										**6.55**	**1**	**0.015**												
cocksfoot																								
fescue				**4.74**	**1**	**0.029[1]**				**5.18**	**1**	**0.029**												
subclovers	**4.18**	**1**	**0.049**																					
no. of different seed	**8.37**	**3**	**0.039[1]**	**8.92**	**3**	**0.030[1]**				**5.52**	**3**	**0.004**	**6.29**	**3**	**0.002**	**4.95**	**3**	**0.007**	**9.04[1]**	**3**	**0.029**	**11.71[1]**	**3**	**0.008**
seed application																								

VARIABLES	centre condition native richness F	DF	P	centre condition exotic richness F	DF	P	centre condition ex:nat richness F	DF	P	centre condition native cover F	DF	P	centre condition exotic cover F	DF	P	centre condition weediness index F	DF	P	edge lifeform cover scat F	DF	P	edge lifeform cover bare ground[1] F	DF	P
years since sown																								
no. of times sown										**5.74**	**1**	**0.022**										**4.29**	**1**	**0.038**
season sown							**3.92[1]**	**1**	**0.048**							**4.01**	**1**	**0.045**						
cocksfoot																								
fescue																								
subclovers																								
no. of different seed																								
seed application																						**7.97**	**1**	**0.005**

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.12. Summary of significant parametric and non-parametric results for sowing variables

	edge lifeform cover native fern			edge lifeform cover native grass[1]			edge lifeform cover native grass %			edge lifeform cover native gram[1]			edge lifeform cover native herb			edge lifeform cover native herb %			edge lifeform cover native shrub			edge lifeform cover exotic grass[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since sown				**11.34**	**3**	**0.010**																		
no. of times sown																								
season sown																								
cocksfoot																								
fescue																						**3.98**	**1**	**0.046**
subclovers																								
no. of different seed													**0.401**	**-**	**0.015**				**4.81**	**3**	**0.036**			
seed application										**4.68**	**1**	**0.030**												

	edge lifeform cover exotic grass %[1]			edge lifeform cover exotic herb			edge lifeform cover exotic herb %[1]			centre lifeform cover scat			centre lifeform cover bare ground			centre lifeform cover native fern[1]			centre lifeform cover native grass[1]			centre lifeform cover native grass %[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since sown																			**8.41**	**3**	**0.038**	**5.42**	**3**	**0.021**
no. of times sown																								
season sown																			**4.04**	**1**	**0.044**			
cocksfoot							**4.07**	**1**	**0.044**															
fescue	**6.50**	**1**	**0.011**													**4.55**	**1**	**0.033**						
subclovers																								
no. of different seed	**10.31**	**3**	**0.016**	**3.43**	**3**	**0.029**										**10.28**	**3**	**0.016**						
seed application																			**4.38**	**1**	**0.036**			

	centre lifeform cover native gram			centre lifeform cover native herb			centre lifeform cover native herb %			centre lifeform cover native shrub			centre lifeform cover exotic grass			centre lifeform cover exotic grass %			centre lifeform cover exotic herb			centre lifeform cover exotic herb %		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
years since sown																								
no. of times sown																								
season sown																								
cocksfoot																								
fescue																								
subclovers																								
no. of different seed				**0.400**	**-**	**0.026**																		
seed application																								

[1]Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

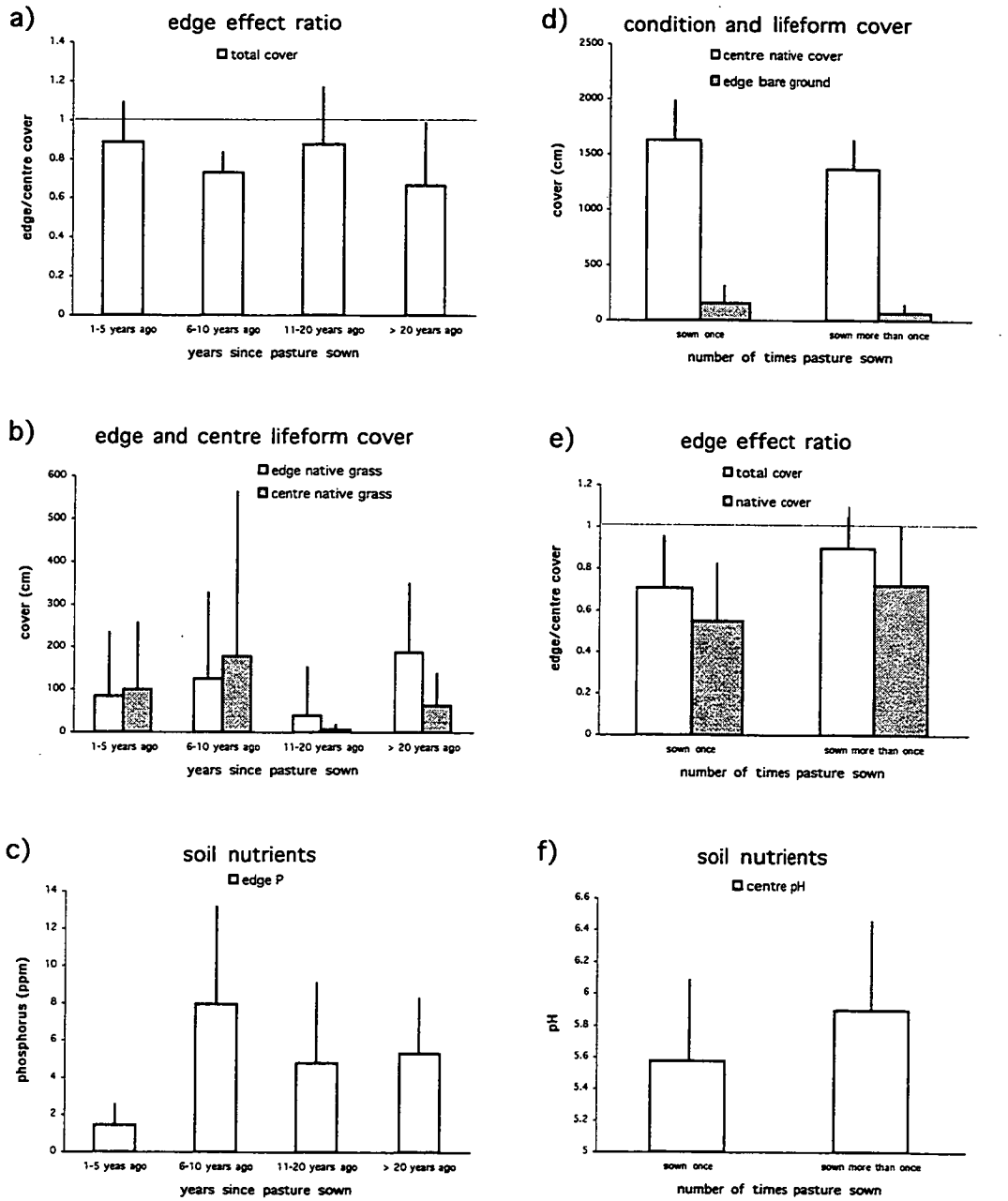


Figure 5.24. Differences in a–c) edge effect ratios, edge and centre lifeform cover, and soil nutrients for the number of years since the pasture was sown, and d–f) condition and lifeform cover, edge effect ratio, and soil nutrients for the number of times the pasture was sown. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation.

regional influences, it can be accepted in this case because the two 'number of times sown' categories were split evenly over the two regions. Also, native cover acted very similarly to total cover, which was not significantly different between the two regions.

One of the reasons that the native cover edge effect ratio was low when the pasture was sown only once was because native cover in the heath centre was higher for this sowing category (Figure 5.24d). The soils in the heath centre were also more acidic when the pasture was sown only once (Figure 5.24f). The sowing of seeds may have been done in

conjunction with fertilising and the spreading of lime which could have drifted to the heath centre. Although, as seen above, liming does not automatically mean the soil will be more alkaline. When the number of times the pasture was sown was tested for the regions, native shrub cover at the edge and fern cover in the centre in the Northeast were higher if sown more than once. The soil nutrients tested give a possible reason, as nitrogen at the edge was higher and pH in the centre was more acidic if sown more than once. The increased native shrub cover coinciding with higher nitrogen levels is consistent with the growth response of natives to minor increases in nutrients.

Only one sowing variable affected the penetration indices and this was the season sown. Autumn sowing had deeper weed penetration compared with other seasons (Figure 5.25a), although this was a small sample — five sites sown in seasons other than autumn. Autumn sowing also had high exotic richness %, exotic/native richness, weediness index and native grass cover (Figure 5.25b–c), all in the heath centre. Exotic richness at the heath edge was also higher in autumn (Figure 5.25d). Although exotic richness is subject to regional influences, it might still be useful because it shows the same pattern as the penetration index and weediness in the heath centre, neither of which varied with region. Soil nutrients again provide an likely explanation for this difference, as the pH in the heath centre was significantly more alkaline for autumn sowing, when there was the higher exotic cover (Figure 5.25e). Another explanation is that two of the sites not sown in autumn were sown in spring and the other three were sown in summer, so perhaps there was a lack of adequate rainfall in these seasons for weed growth.

Pasture seeds were usually applied with a super spreader but also using direct drill or by hand ('other'). There was no response from any of the exotic variables or penetration indices to the method of spreading the seed. Native grasses and graminoids did respond, with higher cover for graminoids at the edge and higher cover for grasses in the centre (Figure 5.25f–g) when a super spreader was used compared with other methods. The native grass result can be ignored as it is subject to regional effects and showed no significant differences when the regions were tested separately. The higher graminoid cover at the edge when a super spreader was used might explain why bare ground at the edge was lower (Figure 5.25f). However, it is unclear why graminoid cover might be higher with the use of a super spreader. In the regions, native richness in the centre was low when a super spreader was used for the region excluding Flinders Island (see note on super regions above). A change in nutrient levels could explain these results, as nitrogen-fixing seeds are used in super spreaders; higher centre nitrogen levels were found with the use of a super spreader (Figure 5.25h).

Many dependent variables responded to the number and type of different seeds used in the sowing mix in the pasture. The presence of Sub-clovers was associated with a higher ratio

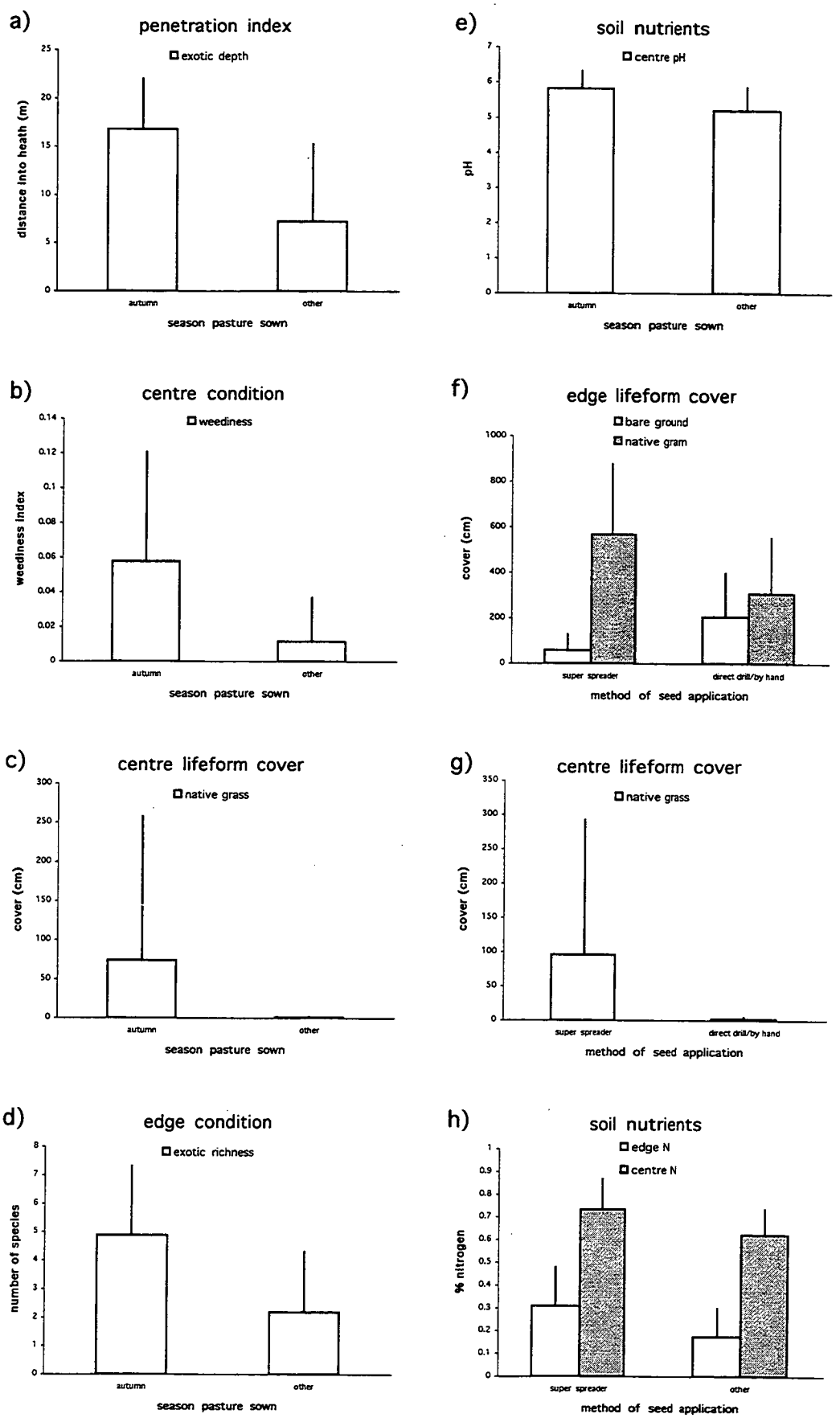


Figure 5.25. Differences in a–e) penetration index, edge and centre condition, centre lifeform cover and soil nutrients for the season the pasture was sown, and f–h) edge and centre lifeform cover, and soil nutrients for the method of seed application. 'Gram', graminoid. Error bars show 1 standard deviation. See text for description of weediness and penetration indices

of edge/centre total and native cover, in the dataset as a whole and in the Northeast region, with the reverse found in the Other region (Figure 5.26a). The soils at the heath edge were more alkaline when Sub-clovers were present, perhaps reflecting an association between sowing and liming (Figure 5.26b). The mechanism for the difference in native cover is not clear, as this result was not found for exotic cover and more alkaline soil would usually encourage exotics rather than natives. It is unlikely that the presence of Sub-clovers in themselves could influence native cover, but it is possible that the use of Sub-clovers coincides with some other environmental or management factor that was not measured.

The same argument holds true for other seeds in the sowing mix. The presence of *Dactylis glomerata* (Cocksfoot) was associated with a significantly higher total richness edge effect ratio (Figure 5.26c), and higher exotic herb % cover at the heath edge. It was also associated with an increase in the amount of native shrub cover at the edge, in the Northeast region. The higher edge effect ratio indicates more species at the edge than the centre, but the significant result was for total species richness not exotic species richness, so it does not necessarily mean that Cocksfoot is more invasive. Again, no soil variations were found that might explain these results, although it is likely that nutrients would be higher where pasture grasses are sown, therefore increasing exotic cover and native growth.

Festuca species (Fescue) was the only pasture species out of the five recorded that was associated with an increase in exotic species. In this case the ratio was well over four times as many exotic species at the edge compared with the centre when Fescue was present (Figure 5.26d). The presence of Fescue also coincided with a higher exotic cover edge effect ratio, higher edge exotic richness, higher edge exotic grass and grass % cover, lower fern cover in the centre (Figure 5.26e–g) and, in the Northeast region, higher native shrub cover. Exotic richness and exotic grass at the edge and native fern at the centre are all subject to regional influences and thus better not used. None of these variables responded to the presence of Fescue when the regions were tested separately. In the Northeast region, it is hard to understand how Fescue in the pasture could influence shrub cover at the heath edge, unless there is some other environmental or management factor influencing both.

Many variables responded to the number of different types of seed in the sowing mix. Differences were apparent for 2 and 4 types of seed compared with 3 and 5 types for: native and exotic richness at the edge, native and exotic richness edge effect ratios, native and exotic cover at the edge, native and exotic cover edge effect ratios, the weediness index edge effect ratio, edge native shrub and exotic herb cover, and centre native fern (Figure 5.27a–h). With some exceptions, generally higher exotic richness and cover were associated with 3 and 5 types of seed and higher native richness and cover associated with

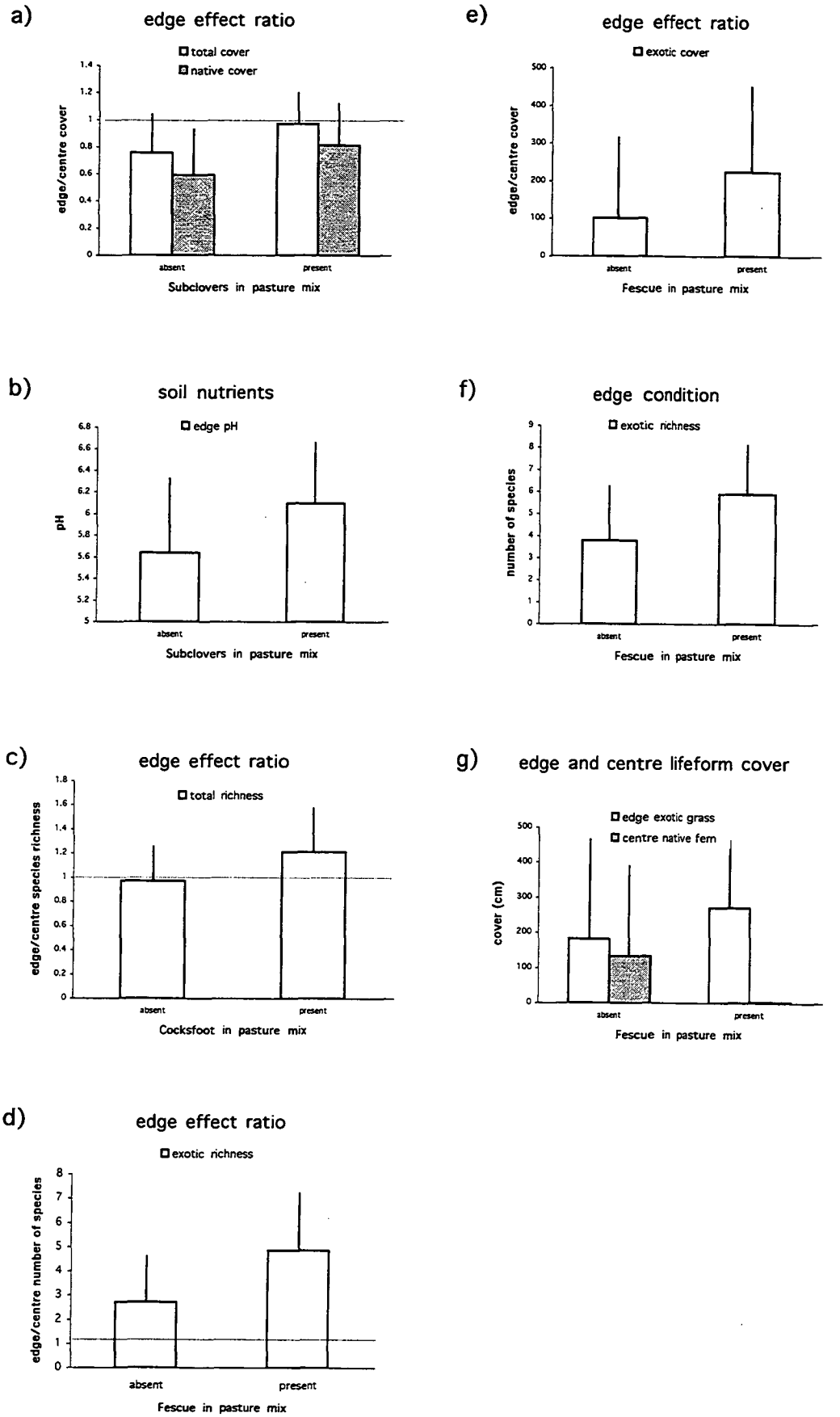


Figure 5.26. Differences in a–b) edge effect ratio and soil nutrients for the presence or absence of Subclovers in the pasture sowing mix, c) edge effect ratio for the presence or absence of Cocksfoot in the pasture sowing mix, and d–g) edge effect ratios, edge condition, and edge and centre lifeform cover for the presence or absence of Fescue in the pasture sowing mix. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation.

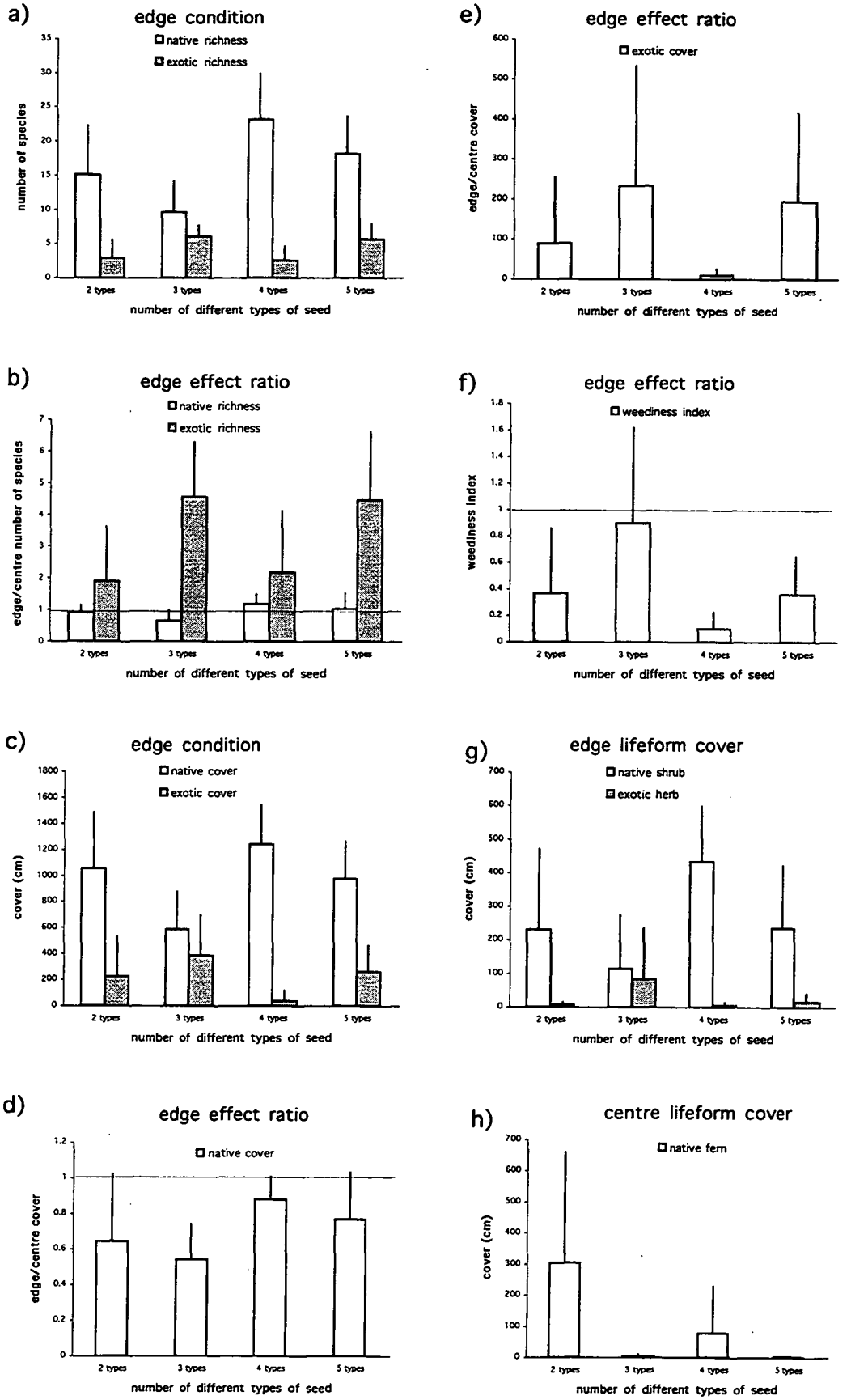


Figure 5.27. Differences in a) edge richness, b) richness edge effect ratio, c) edge cover, d) native cover edge effect ratio, e) exotic cover edge effect ratio, f) weediness index edge effect ratio, g) edge lifeform cover and h) centre lifeform cover. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness index

2 and 4 types of seed. Given the significant results for Fescue, the categories of different numbers of seed were checked for which types of seed they included and excluded. Both 2 and 4 types excluded Fescue, whereas 5 types of seed always included Fescue and 3 types of seed sometimes included Fescue. Whatever environmental or management conditions were governing the influence of Fescue appear also to be at work for the number of different types of seed. The native richness and cover at the edge and native richness and cover edge effect variables can be largely explained by region, because most of the sites with 4 types of seed were in regions that had significantly higher native species richness edge/middle ratios, and about two thirds of the sites with 3 types of seeds sown were in the regions that had lower ratios.

Testing the regions separately showed a more linear response to numbers of seeds in the sowing mix. In the Northeast region, exotic/native richness at the edge and fern cover in the centre decreased as the numbers of seeds increased, whereas native cover and native shrub cover at the edge increased as seed numbers increased. In the Other region, the edge/centre total and native richness ratio also decreased as seed numbers increased. No exotic richness or cover variables responded directly. Again, it is hard to see what effect seed numbers could have on the native richness and cover variables, it is more likely to be that they are both responding to some third factor.

Native mammal trails

Trails made by native mammals from the pasture in the heath had an obvious effect on weed invasion (Table 5.13). There were significant differences in total, native and exotic species richness between trails themselves and the heath to either side of the trails. On the trail, total and native species richness were significantly lower and exotic species richness was significantly higher. The presence of bare ground was significantly related to the trails. Although bare ground was found both on the trail and in the heath to either side, the trails showed significantly more bare ground than the heath. There was significantly lower total and native species richness where there was bare ground but no effect of bare ground on exotic richness.

Table 5.13. Kruskal-Wallis results for total, native and exotic species richness vs native herbivore trails and bare ground (adjusted for ties)

	trail		bare ground	
	H	P	H	P
total species richness	39.11	0.000	30.80	0.000
native species richness	60.36	0.000	27.39	0.000
exotic species richness	23.23	0.000	0.66	0.417

Exotic species richness was also tested against distance along the trails (Figure 5.28). There were significantly more exotics at 2 m compared with 16 and 18 m. Native richness was not significantly different with distance along trail. There were no interactive effects between presence or absence of trail and distance along trail.

These results indicate that native mammals (most noticeably wombats, wallabies and pademelons) can act as vectors to bring exotic species into heathland. The bare ground along the trail seems to play a part in allowing the exotics to establish.

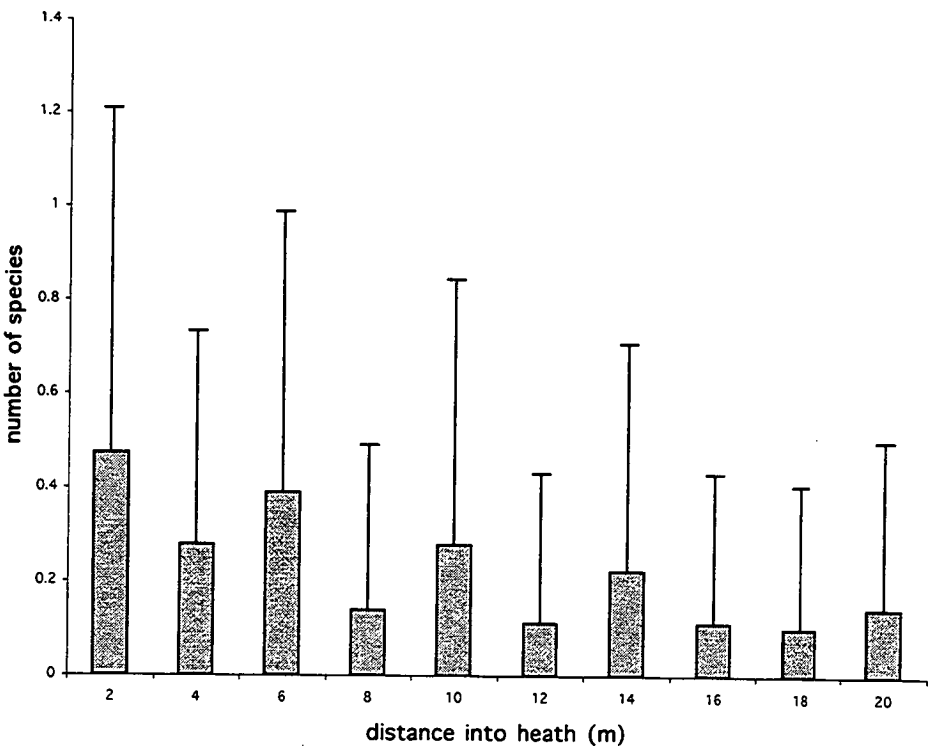


Figure 5.28. Average exotic species richness over distance along native mammal trails. Standard error bars are shown

Heath management

Land tenure

Heathland land tenure generally had a similar effect to pasture land tenure on heathland condition. That is, exotic presence was low for privately owned land and high for parks and reserves (Table 5.14). This could be seen in the lower edge/centre exotic cover ratio for private land, which was due to low edge exotic cover rather than high centre exotic cover (Figure 5.29a, b). This pattern was also seen in low edge exotic species richness, exotic grass cover and exotic herb cover (Figure 5.29c–e). Phosphorus levels at the edge of the heath

Table 5.14. Summary of significant parametric and non-parametric results for heathland management variables

VARIABLES	penetration index exotic depth[1]			penetration index exotic richness			penetration index exotic cover			penetration index exotic weediness			edge effect ratio total richness			edge effect ratio native richness			edge effect ratio exotic richness			edge effect ratio total cover		
	H	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
land tenure																						3.51	3	0.022
last fire																						3.46	2	0.038
heath cleared													6.82	1	0.013							12.84	1	0.001
heath slashed																			8.27	1	0.006			
where cleared							4.77	2	0.016				12.09	2	0.000	4.27	2	0.021				4.86	2	0.015
current grazing																								
previous grazing																3.52	2	0.040						
type of stock																								
season stock grazed													4.82	2	0.014							6.88	2	0.003

	edge effect ratio native cover			edge effect ratio exotic cover			edge condition native richness			edge condition exotic richness			edge condition ex:nat richness			edge condition native cover			edge condition exotic cover			edge condition weediness index		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
land tenure	4.86	3	0.005	11.34	3	0.010[1]				4.14	3	0.011				3.48	3	0.023	9.31[1]	3	0.025			
last fire	5.51	2	0.007													3.95	2	0.025						
heath cleared	11.36	1	0.002										5.94	1	0.019									
heath slashed										7.24	1	0.010							4.31[1]	1	0.038	6.19[1]	1	0.013
where cleared	5.62	2	0.007	6.69	2	0.035[1]										4.64	2	0.016						
current grazing							3.83	2	0.031				4.68	2	0.015									
previous grazing							4.50	2	0.018															
type of stock																								
season stock grazed	4.08	2	0.025																					

	centre condition native richness			centre condition exotic richness			centre condition ex:nat richness			centre condition native cover			centre condition exotic cover			centre condition weediness index			edge lifeform cover scat[1]			edge lifeform cover bare ground[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
land tenure	3.61	3	0.021																			9.20	3	0.027
last fire																								
heath cleared	6.67	1	0.014							4.56	1	0.040												
heath slashed																								
where cleared	4.50	2	0.019																					
current grazing																								
previous grazing	3.46	2	0.044																					
type of stock	4.91	2	0.014																6.97	2	0.031			
season stock grazed	6.40	2	0.005							5.28	2	0.011							6.58	2	0.037			

[1] Results for Kruskal-Wallis rank order test

Table 5.14. Summary of significant parametric and non-parametric results for heathland management variables

	edge lifeform cover native fern			edge lifeform cover native grass[1]			edge lifeform cover native grass %			edge lifeform cover native gram			edge lifeform cover native herb			edge lifeform cover native herb %[1]			edge lifeform cover native shrub			edge lifeform cover exotic grass[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
land tenure																			**3.97**	**3**	**0.020**	**10.16**	**3**	**0.017**
last fire																								
heath cleared																								
heath slashed																								
where cleared																**9.28**	**2**	**0.010**						
current grazing																			**5.77**	**2**	**0.015**			
previous grazing																								
type of stock				**7.22**	**2**	**0.027**																		
season stock grazed																								

	edge lifeform cover exotic grass %			edge lifeform cover exotic herb			edge lifeform cover exotic herb %[1]			centre lifeform cover scat			centre lifeform cover bare ground			centre lifeform cover native fern[1]			centre lifeform cover native grass			centre lifeform cover native grass %[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
land tenure				**3.58**	**3**	**0.020**	**16.90**	**3**	**0.001**				**4.66**	**3**	**0.007**							**8.42**	**3**	**0.038**
last fire																								
heath cleared							**4.76**	**1**	**0.029**				**6.84**	**1**	**0.013**									
heath slashed				**4.92**	**1**	**0.032**	**6.81**	**1**	**0.009**							**8.52**	**1**	**0.004**						
where cleared				**3.70**	**2**	**0.034**	**11.56**	**2**	**0.003**							**6.83**	**2**	**0.033**						
current grazing				**5.00**	**2**	**0.012**	**8.04**	**2**	**0.018**															
previous grazing				**6.16**	**2**	**0.005**	**10.82**	**2**	**0.004**															
type of stock																								
season stock grazed													**4.03**	**2**	**0.028**									

	centre lifeform cover native gram			centre lifeform cover native herb[1]			centre lifeform cover native herb %			centre lifeform cover native shrub[1]			centre lifeform cover exotic grass			centre lifeform cover exotic grass %			centre lifeform cover exotic herb			centre lifeform cover exotic herb %		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
land tenure	**3.55**	**3**	**0.022**							**8.31**	**3**	**0.040**												
last fire				**6.28**	**2**	**0.043**																		
heath cleared										**8.44**	**1**	**0.004**												
heath slashed																								
where cleared																								
current grazing																								
previous grazing																								
type of stock																								
season stock grazed																								

[1]Results for Kruskal-Wallis rank order test

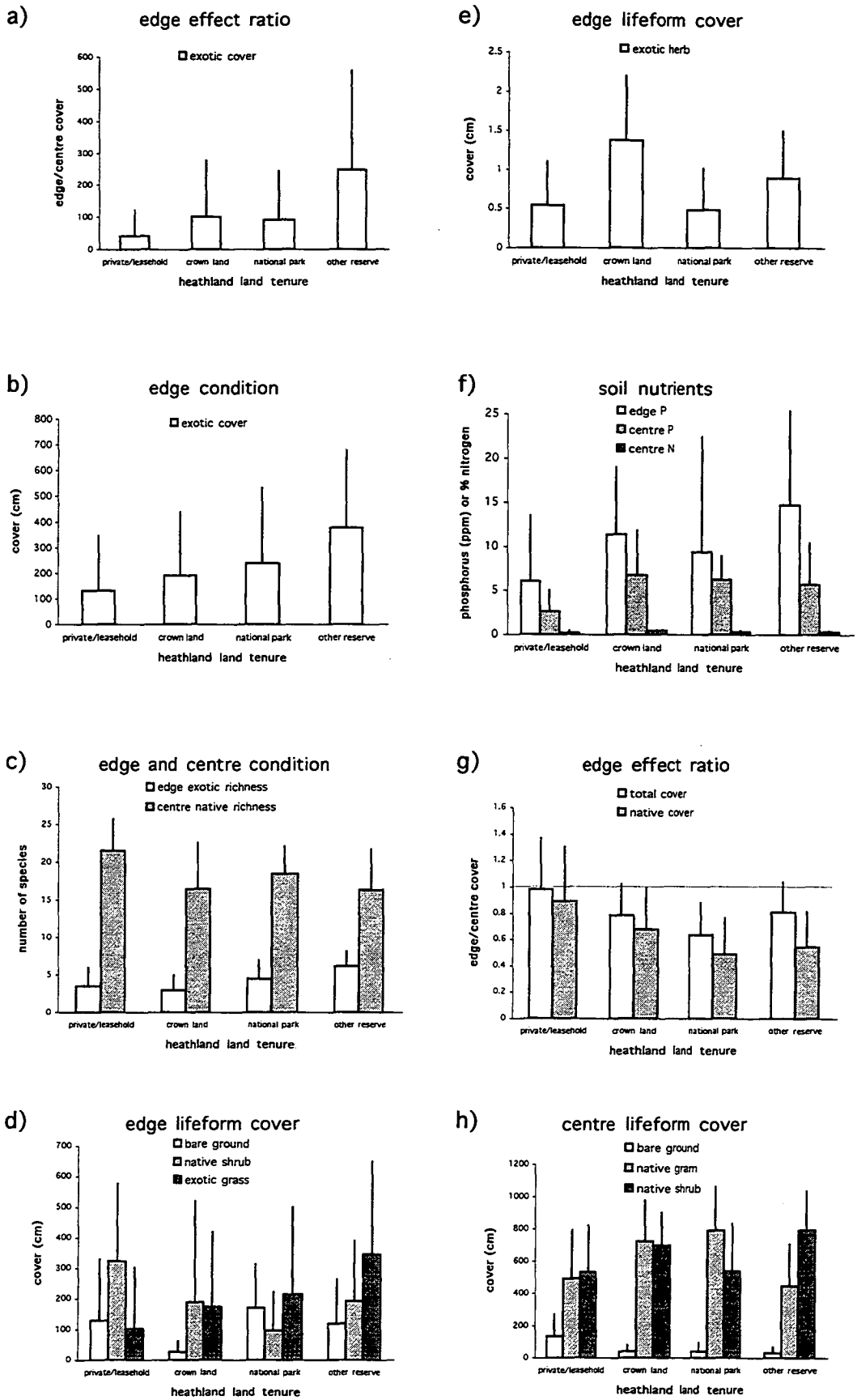


Figure 5.29. Differences in a) edge effect ratio, b–c) edge and centre condition, d–e) edge lifeform cover, f) soil nutrients, g) edge effect ratio and h) centre lifeform cover for heathland land tenure. 'Gram', graminoid. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation.

were also low for private land, and it is quite possible that this was affecting the weed invasion (Figure 5.29f). No exotic variables in the heath centre responded to land tenure. Many of the variables showed regional differences, and the land tenure categories were not split evenly over the regions: there was more private heathland in the Other region and more 'other reserve' heathland in the Northeast region. Testing the regions separately only resulted in one exotic variable responding to land tenure: edge exotic herb % cover in the Northeast was lowest for private land and highest for crown land, the same pattern as seen in the whole dataset. Despite the complication of regional influences there does seem to be a pattern of low weed presence for privately owned land, where one might expect there to be a greater level of disturbance, and higher weed presence in parks and reserves.

For the native condition and lifeform variables, there was the opposite pattern to the exotic response. Private heathland tended to have a more distinct boundary compared with crown land and national park, as shown by the edge/centre native cover ratio (Figure 5.29g). This was due to higher cover of native shrubs at the heath edge and lower native graminoid cover in the centre (Figure 5.29d, h). Increased growth of heath plants at the edge could be due to a 'hedge effect', in response to increased fertilisers from the pasture, but the soil nutrient measurements do not support this theory, as phosphorus levels were low for private land (Figure 5.29f). A more diffuse edge for reserved land was also seen in the Northeast region, as shown by a low edge/centre native cover ratio for national parks. In the centre of the heathland, native richness was lower for reserves and crown land than it was for private land (Figure 5.29c), but there was high native shrub cover for other reserves, whereas bare ground was low (Figure 5.29h). It is speculated that soil nutrients in the centre could be affecting the native heathland plants, with greater richness in private land where phosphorus was low but higher native shrub cover in crown land where nitrogen levels were also high (Figure 5.29f). It is beyond the scope of this study to ascertain whether the current or past management of the national parks and other reserves has affected the species diversity of the heathlands, or if they were already lower prior to the reserves being established. Most of the native variables were not significantly different between the different regions, so can be accepted for the whole dataset. The exception is native shrub cover at the edge, but this variable did not respond when tested for each region separately. In the East Coast and Waterhouse areas, some management by private landowners must have been conducive to high heathland diversity at the boundary. The same was the case for all areas excluding Flinders Island.

Fire history

The ratio of edge to centre native cover declined over time since the last burn in the heathland (Figure 5.30a). This decline was due to decreasing cover at the edge of the

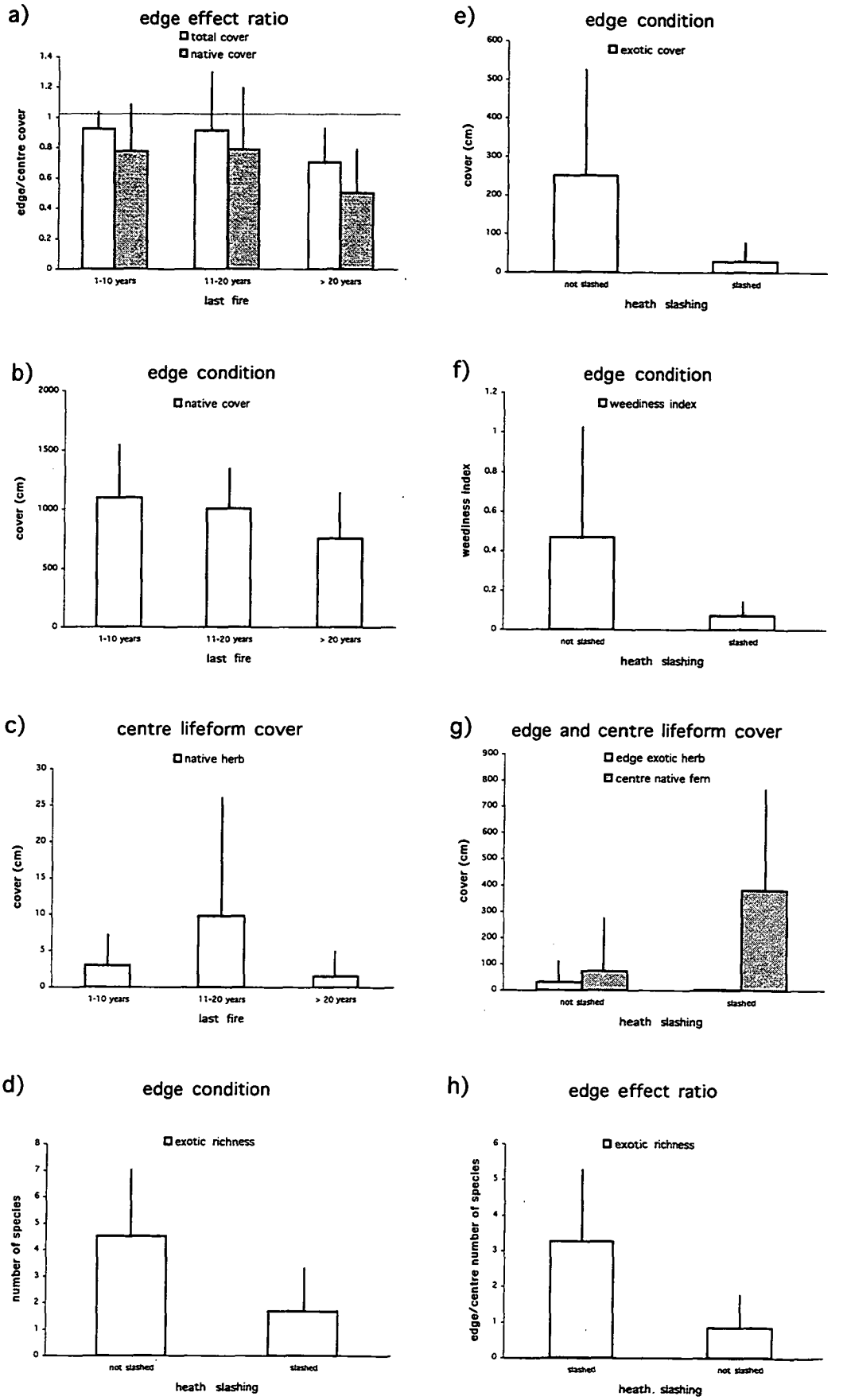


Figure 5.30. Differences in a–c) edge effect ratio, edge condition and centre lifeform cover for time since last fire, and d–h) edge condition, edge and centre lifeform cover, and edge effect ratio for presence or absence of slashing in the heath. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness index

heath (Figure 5.30b) and a relative increase in the heath centre. In the Northeast region, the edge/centre native cover ratio and native cover at the edge responded to time since fire in the same way as described above. Native herbs were highest for 11–20 years and very low for more than 20 years (Figure 5.30c). This is not surprising for native herbs, which would become easily shaded out as shrub cover increased with time since fire.

Clearing and slashing

For both slashing and clearing, although there were generally less exotics at the edge, more exotics were able to penetrate deeper into the heath. Exotic richness, exotic/native richness, exotic cover, weediness index, exotic herb and herb % cover at the heath edge were all lower if slashed (Figure 5.30d–g). Regular slashing in the heath was associated with more exotics in the interior of the heath compared with the edge, whereas no slashing meant about four times as many exotics at the edge than the interior (Figure 5.30h). In addition, clearing inside the heath had a much higher penetration of exotic cover (and lower ratio of edge to centre exotic cover) compared with clearing along the boundary or no clearing at all (Figure 5.31a, b). In particular, exotic herb cover at the edge was higher for clearing at the boundary or inside the heath (Figure 5.31c). Clearing is a type of disturbance usually associated with weed invasion, so its occurrence well within the heath and not just along the edge means more opportunities for weeds to become established.

Native plant cover and richness were also affected by whether or not the heath was slashed, whether it was cleared and where it was cleared. If the heathland was cleared, there was a higher total and native cover edge effect ratio, due to lower cover in the heath centre, specifically low native shrub cover and higher bare ground (Figure 5.32a–b). It makes sense that there would be lower cover, particular of shrubs, when some sort of clearing had occurred. However, when the site of the clearing was tested, the native cover edge effect ratio was also lowest when clearing occurred inside the heath, but in this case it was due to relatively higher cover in the heath centre, which is the reverse of what one might expect (Figure 5.31d, e). Clearing along the boundary was associated with a sharper boundary, with equal total cover at edge and centre (Figure 5.31d). Fern cover in the centre was higher if slashed or cleared, but this probably almost entirely due to the response of bracken to slashing (Figures 5.30g and 5.31c).

The ratio of total species richness between edge and centre was lower when clearing was present, and this relative difference was due to higher native richness in the heath centre (Figure 5.32c, d). Soil nutrients also varied with clearing, the level of phosphorus at the edge being higher if the heath was not cleared (Figure 5.32e). If clearing was associated

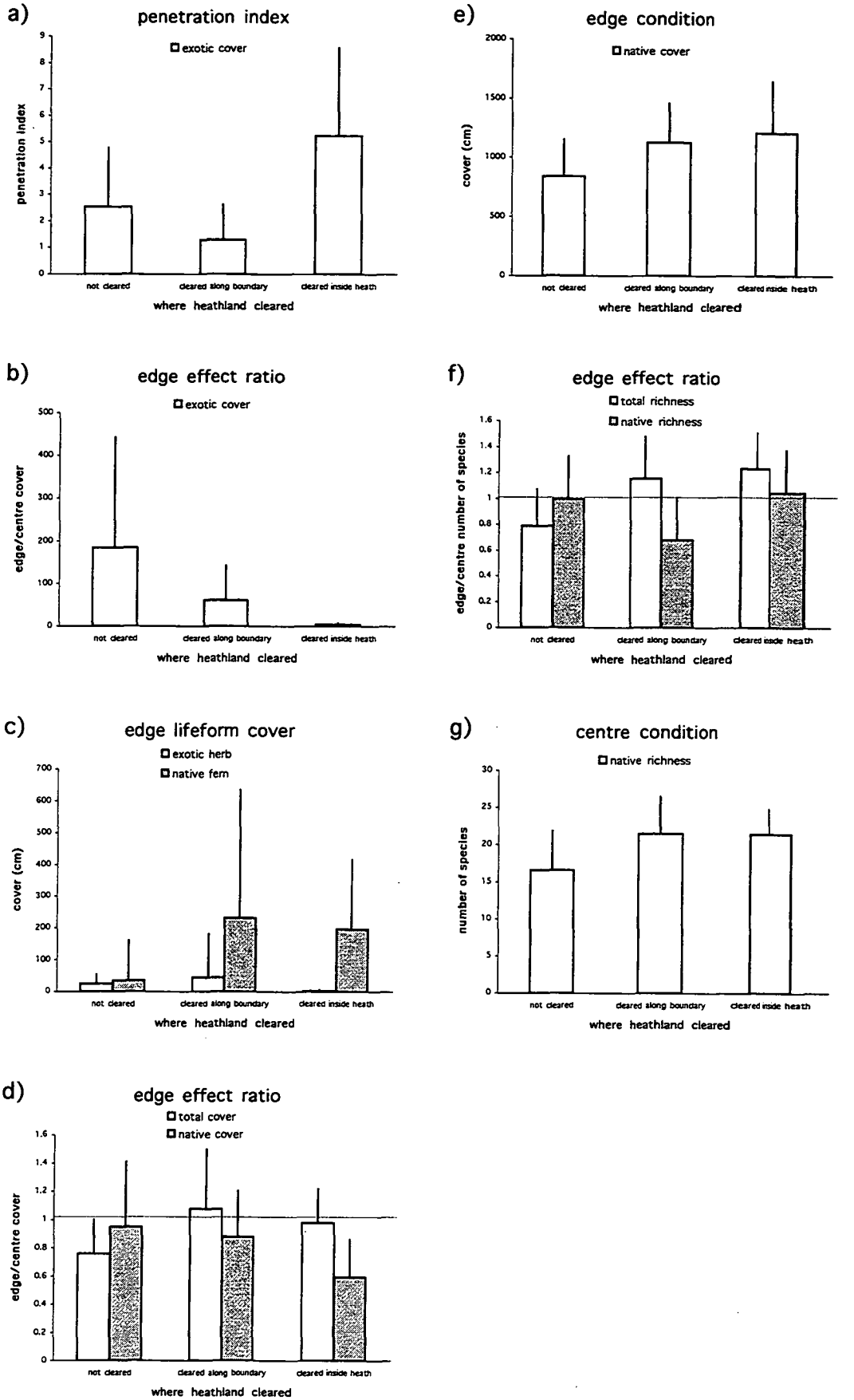


Figure 5.31. Differences in a) penetration index, b) exotic cover edge effect ratio, c) edge lifeform cover, d) total and native cover edge effect ratio e) edge condition, f) richness edge effect ratio and g) centre condition for where the heathland was cleared. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of penetration index

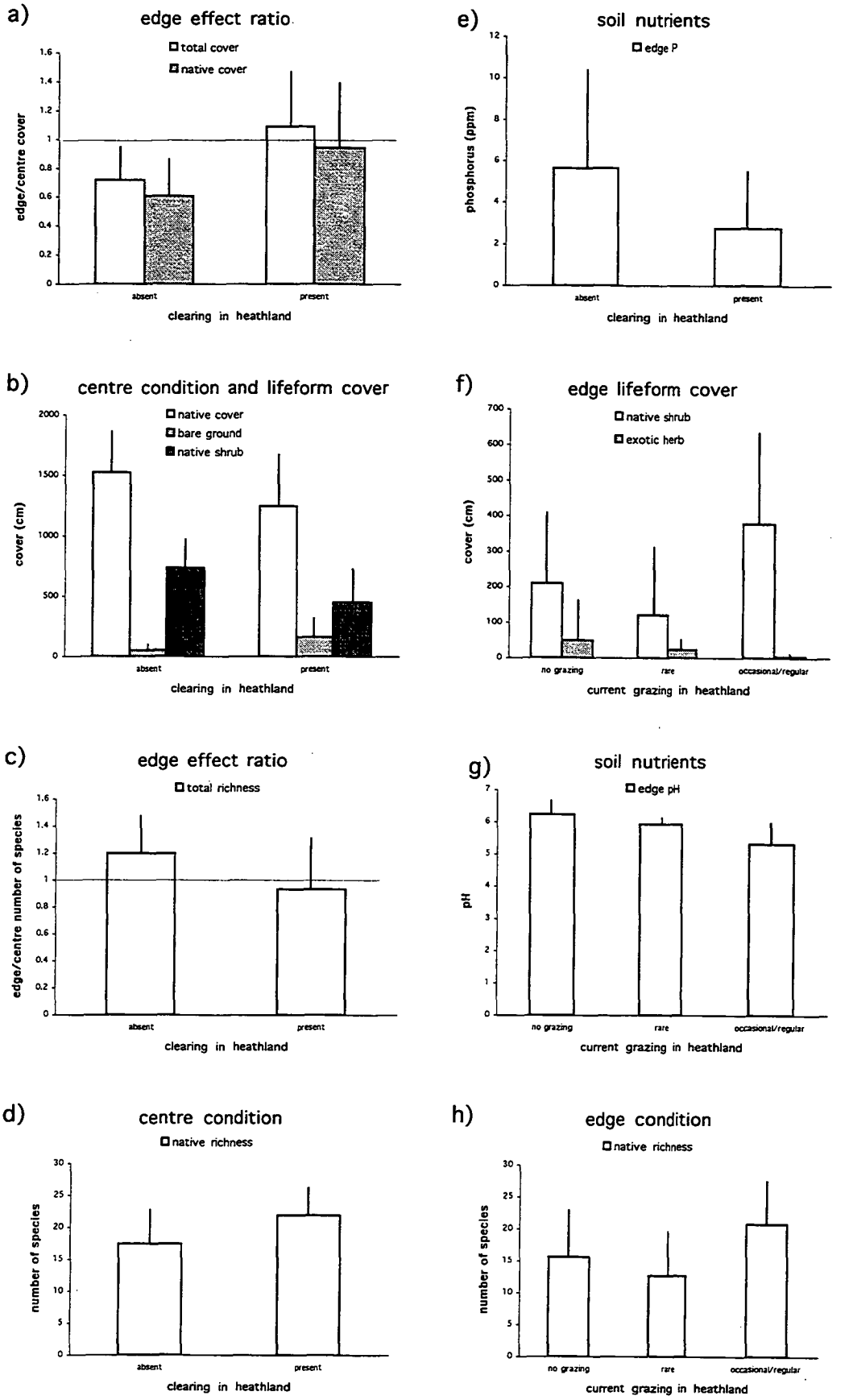


Figure 5.32. Differences in a–e) edge effect ratios, centre condition, centre lifeform cover and soil nutrients for presence or absence of clearing in the heathland, and f–h) edge lifeform cover, soil nutrients and edge condition for the level of current grazing in the heathland. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation.

with pasture management, particularly fertilisation, one would expect the phosphorus level to be reversed, which would also make more sense given the species richness result, but this was not the case. Clearing within the heath increased species richness in the centre but clearing along the boundary reduced species richness at the edge, as shown by the edge effect ratio and centre richness (Figure 5.31f, g). The increased species richness at the edge could be due to light-loving or ephemeral species taking advantage of the conditions. The decreased native species richness at the edge could be due to the increased competition from exotic species or conditions at the edge being unfavourable.

Heathland grazing

Domestic stock grazing in the heath was divided into current grazing, previous grazing, the type of stock and the time of year the grazing occurred. The level of current grazing affected exotic herb cover, exotic herb % cover and exotic/native richness at the edge, with them declining as grazing increased (Figure 5.32f). The decreasing cover could have been due to decreasing pH at the edge (Figure 5.32g). Increasing presence of domestic stock therefore appears to make the soil more acidic. When the presence or absence of stock grazing was scored as a disturbance for each quadrat, exotic richness was lower and bare ground was higher at the edge when stock were present. There were no differences in soil pH but it is speculated that the same mechanism could be occurring. Native richness and shrub cover at the edge also tended to increase as stock grazing increased (Figure 5.32f, h). All the variables that responded to current grazing were affected by region. When tested separately, edge soil pH in both the Northeast and the Other regions showed the same pattern of increasing acidity with increasing grazing pressure. In the Other region, nitrogen levels also increased.

The native richness edge effect ratio decreased as the time since the heath was grazed increased, due to both lower edge richness and higher centre richness (Figure 5.33a, b). Domestic stock could have roamed over the entire heathland area, so they could have affected both edge and centre richness. However, grazing does not appear to have harmed native species richness in the centre, as it was lowest when no grazing had occurred. Recent grazing was associated with the lowest level of exotic herb cover, whereas grazing more than 5 years ago was associated with the highest level (Figure 5.33c). Exotic herb cover was subject to regional influences, but there was a similar pattern when tested for the Northeast region, that is, exotic herb % cover and exotic richness % were highest when the heath had been grazed more than 5 years ago.

The type of stock did not appear to have much of an impact on heath condition or weed invasion, but it was only really possible to divide the stock type into cattle alone or 'other'

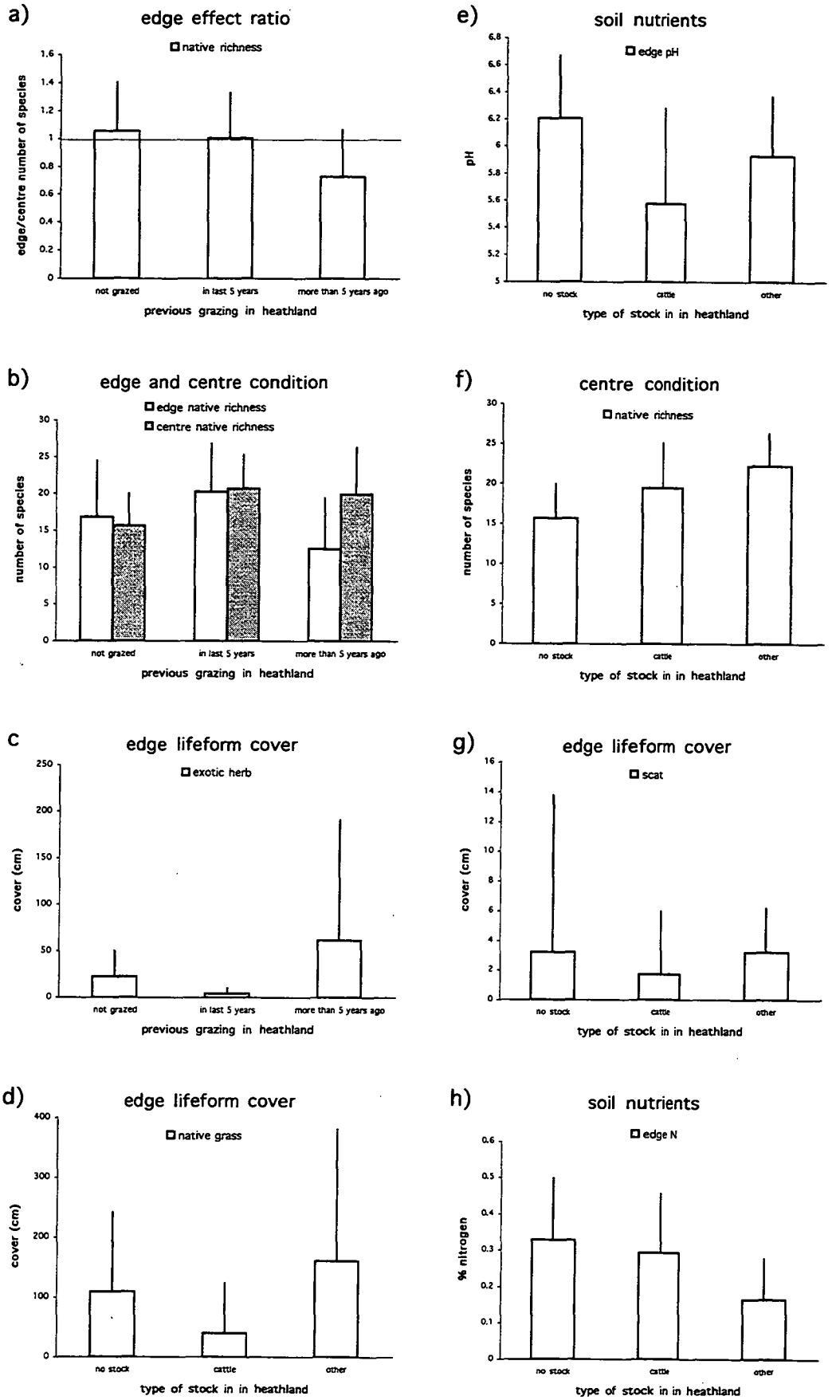


Figure 5.33. Differences in a–c) edge effect ratio, edge and centre condition and edge lifeform cover for previous grazing in the heathland, and d–h) edge lifeform cover, soil nutrients and centre condition for the type of stock grazing in the heathland. Horizontal line indicates equal amount of species richness between heath edge and centre. Error bars show 1 standard deviation.

(sheep, sheep with cattle, pigs, camels with cattle). Cattle seemed to preferentially graze native grass, whereas other stock seemed to encourage native grass (Figure 5.33d). Soil pH was also more acidic at the edge when cattle were present and this was true for the Other region as well (Figure 5.33e). However, it could be that the stock type was related to another environmental variable that was influencing native grass cover. Other stock had more of an effect on native species richness and scat cover at the edge, both of which were highest when they were present (Figure 5.33f, g). Nitrogen levels were also lowest for other stock (Figure 5.33h).

Grazing by stock all year meant less total species but more total and native cover (increased dominance by fewer species) at the edge than the centre, whereas grazing part of the year was not significantly different to no grazing, both of which had more species but less cover at the edge than the centre (increased polydominance) (Figure 5.34a, b). In the Northeast region, the same was true for edge/centre native cover. The increased polydominance was due to more richness and less cover in the centre (Figure 5.34c, d). When stock were able to graze in the heath all year, scats at the edge were higher, native richness and bare ground in the centre were higher, and the soils were more acidic (Figure 5.34e, f).

General disturbance

Fences, ditches and tracks generally acted as barriers to weed invasion (Table 5.15). The depth of penetration of weeds decreased when there were two fences at the boundary compared with one or none (Figure 5.35a). In the heath centre, exotic richness, exotic cover, the weediness index and native herb cover decreased as the number of fences increased (Figure 5.35b–e). No soil differences that could influence this result were present in the centre, although the pH at the edge was more acidic for no fences (Figure 5.35f). Given the results described so far, one would assume that there would also be less exotics at the edge for one or two fences. However, the opposite was the case, with exotic richness and herb % cover highest when there was one fence (Figure 5.35b). Exotic richness in the Northeast region behaved in the same way as for the entire dataset, that is, exotic richness at the edge was highest for once fence. There is obviously a relationship between fencing and domestic stock: stock presence leads to a need for fencing and also increases the likelihood of weeds, no stock meaning less exotics and also no fences. Scat cover was high at the edge of the heath when there was no fence and low for one or two fences (Figure 5.35e).

When a track was present, exotic species richness at the heath edge was low, indicating the track acting as a barrier rather than a conduit for weed species (Figure 5.35g). The depth of penetration, exotic richness penetration index, weediness index, and exotic richness at the edge and centre were also reduced if there was a ditch at the boundary

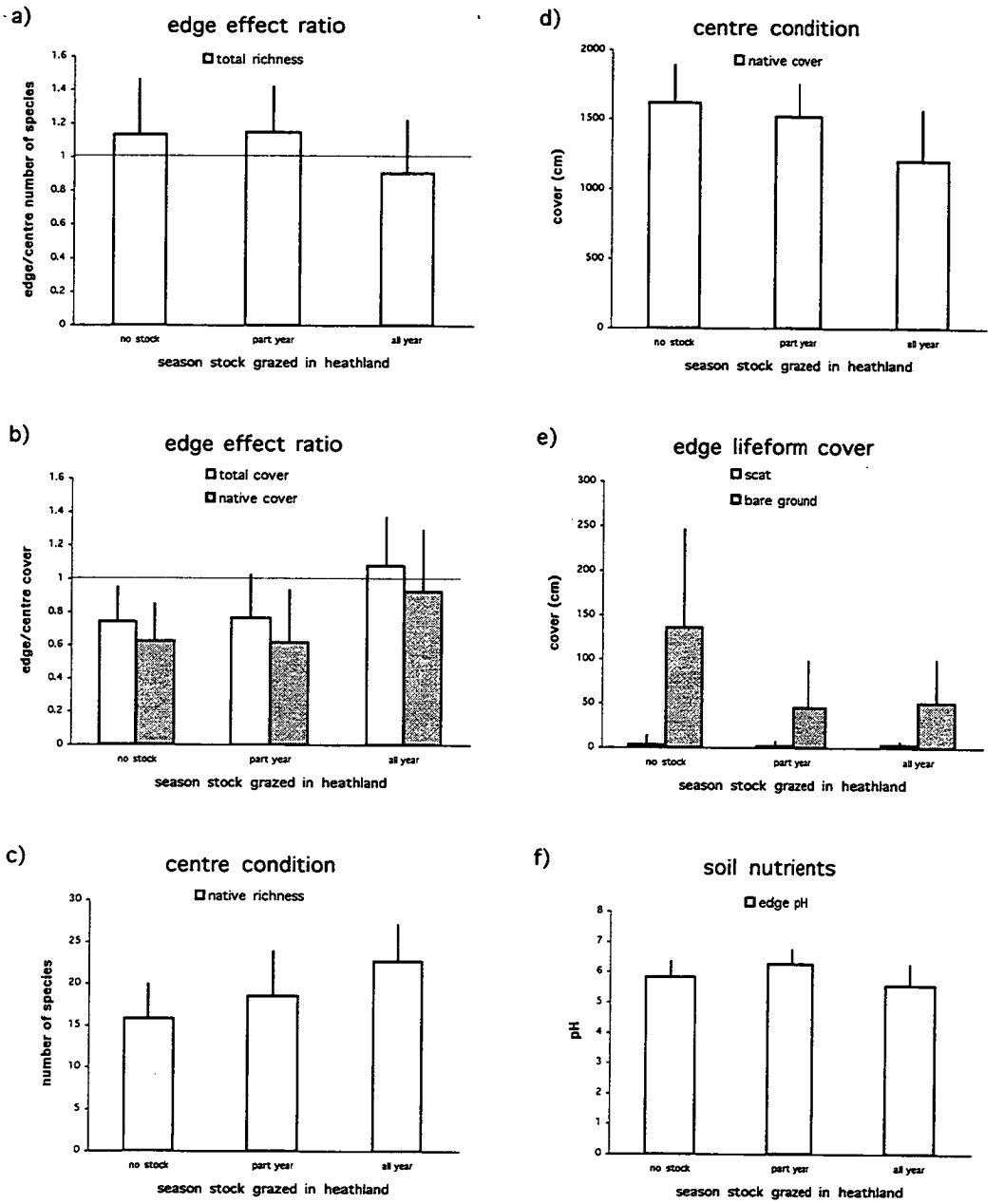


Figure 5.34. Differences in a–b) edge effect ratio, c–d) centre condition, e) edge lifeform cover and f) soil nutrients for the season stock grazed in the heathland. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation.

compared with no ditch (Figure 5.36a–c). The presence of a ditch at the boundary was also associated with more acidic soil in the heath centre (Figure 5.36d). This could be an indication of the presence of wet heath communities which occur on soils that are more poorly drained (thus requiring artificial drainage) and more acidic (Kirkpatrick and Harris 1999b).

Table 5.15. Summary of significant parametric and non-parametric results for heathland disturbance variables

VARIABLES	penetration index exotic depth[1]			penetration index exotic richness			penetration index exotic cover			penetration index exotic weediness			edge effect ratio total richness			edge effect ratio native richness			edge effect ratio exotic richness			edge effect ratio total cover		
	H	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
fence	**8.75**	**2**	**0.013**																			**3.70**	**2**	**0.031**
track																								
ditch	**6.51**	**1**	**0.011**	**5.99**	**1**	**0.018**				**4.16**	**1**	**0.047**												
boundary index																						**5.48**	**3**	**0.002**
disturbance index edge																			**2.93**	**3**	**0.042**			
disturbance index B													**4.71**	**3**	**0.005**	**3.30**	**3**	**0.027**						
disturbance index C													**5.10**	**3**	**0.045**	**4.88**	**3**	**0.004**						
disturbance index D													**3.88**	**3**	**0.014**									
disturbance index centre																						**7.32**	**2**	**0.002**
edge/centre disturbance													**0.323**	**+**	**0.016**							**0.395**	**-**	**0.003**
P. cinn edge	**6.43**	**1**	**0.011**																					
P. cinn centre	**5.68**	**1**	**0.017**	**9.77**	**1**	**0.003**																		

	edge effect ratio native cover			edge effect ratio exotic cover			edge condition native richness			edge condition exotic richness			edge condition ex:nat richness			edge condition native cover			edge condition exotic cover			edge condition weediness index		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
fence										**4.69**	**2**	**0.013**												
track										**4.68**	**1**	**0.035**												
ditch										**4.71**	**1**	**0.034**												
boundary index																								
disturbance index edge	**4.52**	**3**	**0.041**	**3.20**	**3**	**0.032**										**5.38**	**3**	**0.003**				**10.25**[1]	**3**	**0.017**
disturbance index B							n/a			n/a			n/a			n/a			n/a			n/a		
disturbance index C							n/a			n/a			n/a			n/a			n/a			n/a		
disturbance index D							n/a			n/a			n/a			n/a			n/a			n/a		
disturbance index centre							n/a			n/a			n/a			n/a			n/a			n/a		
edge/centre disturbance			**0.005-**			**0.004-**	n/a			n/a			n/a			n/a			n/a			n/a		
animal diggings																			**5.67**[1]	**1**	**0.017**			
windpruning							**4.13**	**1**	**0.047**				**4.38**	**1**	**0.041**	**7.05**	**1**	**0.010**	**5.18**[1]	**1**	**0.023**	**5.89**[1]	**1**	**0.015**
soil disturbance							**4.44**	**1**	**0.040**															
stock grazing										**6.98**	**1**	**0.011**	**4.43**	**1**	**0.040**									
P. cinn edge	**5.03**	**1**	**0.029**							**4.30**	**1**	**0.043**	**5.15**	**1**	**0.028**							**3.93**[1]	**1**	**0.048**
P. cinn centre										n/a			n/a			n/a			n/a			n/a		

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.15. Summary of significant parametric and non-parametric results for heathland disturbance variables

	centre condition native richness			centre condition exotic richness			centre condition ex:nat richness			centre condition native cover			centre condition exotic cover[1]			centre condition weediness index			edge lifeform cover scat[1]			edge lifeform cover bare ground		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
fence				7.85[1]	2	0.020	7.28[1]	2	0.026				6.50	2	0.039	7.49	2	0.024	10.04	2	0.007			
track																								
ditch				4.30[1]	1	0.038																		
boundary index																								
disturbance index edge	n/a			n/a			n/a			n/a			n/a			n/a			10.60	3	0.014			
disturbance index centre	5.50	2	0.007							5.79	2	0.006							n/a			n/a		
native grazing	14.00	1	0.000																					
fallen branches													7.55	1	0.006									
other disturbance										4.08	1	0.049												
P. cinn edge	n/a			n/a			n/a			n/a			n/a			n/a								
P. cinn centre				5.91[1]	1	0.015	4.70[1]	1	0.030				5.12	1	0.024	4.74	1	0.029	n/a			n/a		

	edge lifeform cover native fern			edge lifeform cover native grass[1]			edge lifeform cover native grass %			edge lifeform cover native gram[1]			edge lifeform cover native herb			edge lifeform cover native herb %[1]			edge lifeform cover native shrub			edge lifeform cover exotic grass[1]		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
fence													6.03	2	0.049	6.19	2	0.045						
track																								
ditch																								
boundary index																								
disturbance index edge																								
native grazing				5.27	1	0.022																		
animal diggings																						6.41	1	0.011
wind pruning																						5.24	1	0.022
other disturbance										4.38	1	0.036												
P. cinn edge																								

[1] Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

Table 5.15. Summary of significant parametric and non-parametric results for heathland disturbance variables

	edge lifeform cover exotic grass %[1]			edge lifeform cover exotic herb			edge lifeform cover exotic herb %[1]			centre lifeform cover scat			centre lifeform cover bare ground			centre lifeform cover native fern			centre lifeform cover native grass[1]			centre lifeform cover native grass %		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
fence							**9.95**	**2**	**0.007**													**4.65**	**1**	**0.031**
track																						**8.35**	**3**	**0.039**
ditch										n/a			n/a			n/a			n/a			n/a		
boundary index	**8.06**	**3**	**0.045**																					
disturbance index edge				n/a			n/a						**3.84**	**2**	**0.028**									
disturbance index centre	n/a																							
animal diggings	**4.00**	**1**	**0.046**										**6.01**	**1**	**0.018**									
stock grazing																						**6.88**	**1**	**0.009**
native grazing										n/a			n/a			n/a			n/a			n/a		
P. cinn edge																								
P. cinn centre	n/a			n/a			n/a																	

	centre lifeform cover native gram			centre lifeform cover native herb[1]			centre lifeform cover native herb %[1]			centre lifeform cover native shrub			centre lifeform cover exotic grass[1]			centre lifeform cover exotic grass %[1]			centre lifeform cover exotic herb			centre lifeform cover exotic herb %		
	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P	F	DF	P
fence				**6.87**	**2**	**0.032**																		
track																								
ditch				**5.30**	**1**	**0.021**	**8.63**	**1**	**0.015**															
boundary index																								
disturbance index centre				**8.10**	**2**	**0.017**	**6.33**	**2**	**0.010**															
fallen branches													**6.27**	**1**	**0.012**	**6.98**	**1**	**0.008**						
animal diggings																			**4.08**	**1**	**0.043**			
P. cinn centre																			**5.28**	**1**	**0.022**	**5.42**	**1**	**0.020**

[1]Results for Kruskal-Wallis rank order test. +, positive correlation; -, negative correlation; Pearson's Product Moment Correlation Co-efficient to left of sign; P value to right of sign

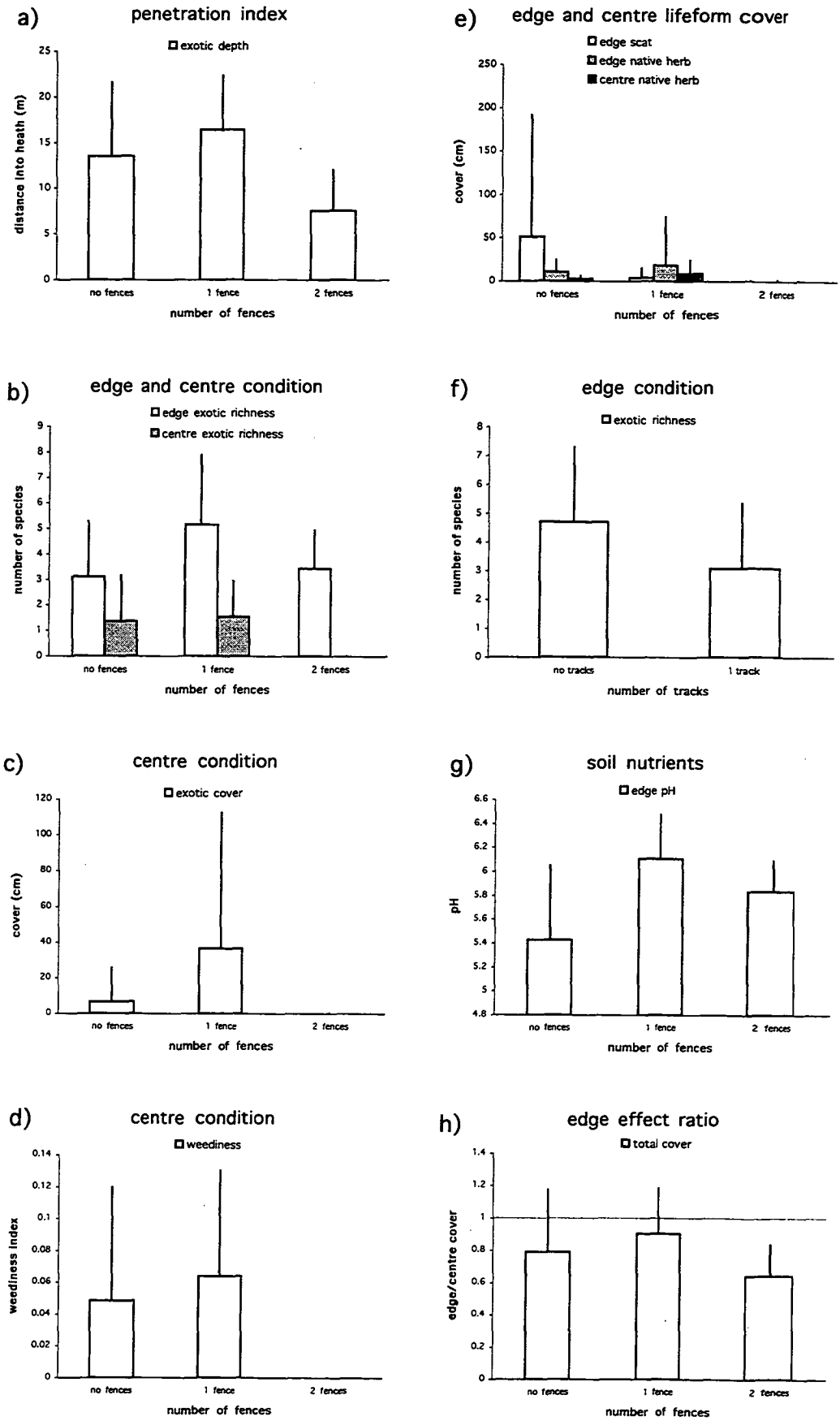


Figure 5.35. Differences in a) penetration index, b–d) edge and centre condition, e) edge and centre lifeform cover, f) exotic richness, g) soil nutrients and h) edge effect ratio for the number of fences at the heath–pasture boundary. Horizontal line indicates equal amount of cover richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness and penetration indices

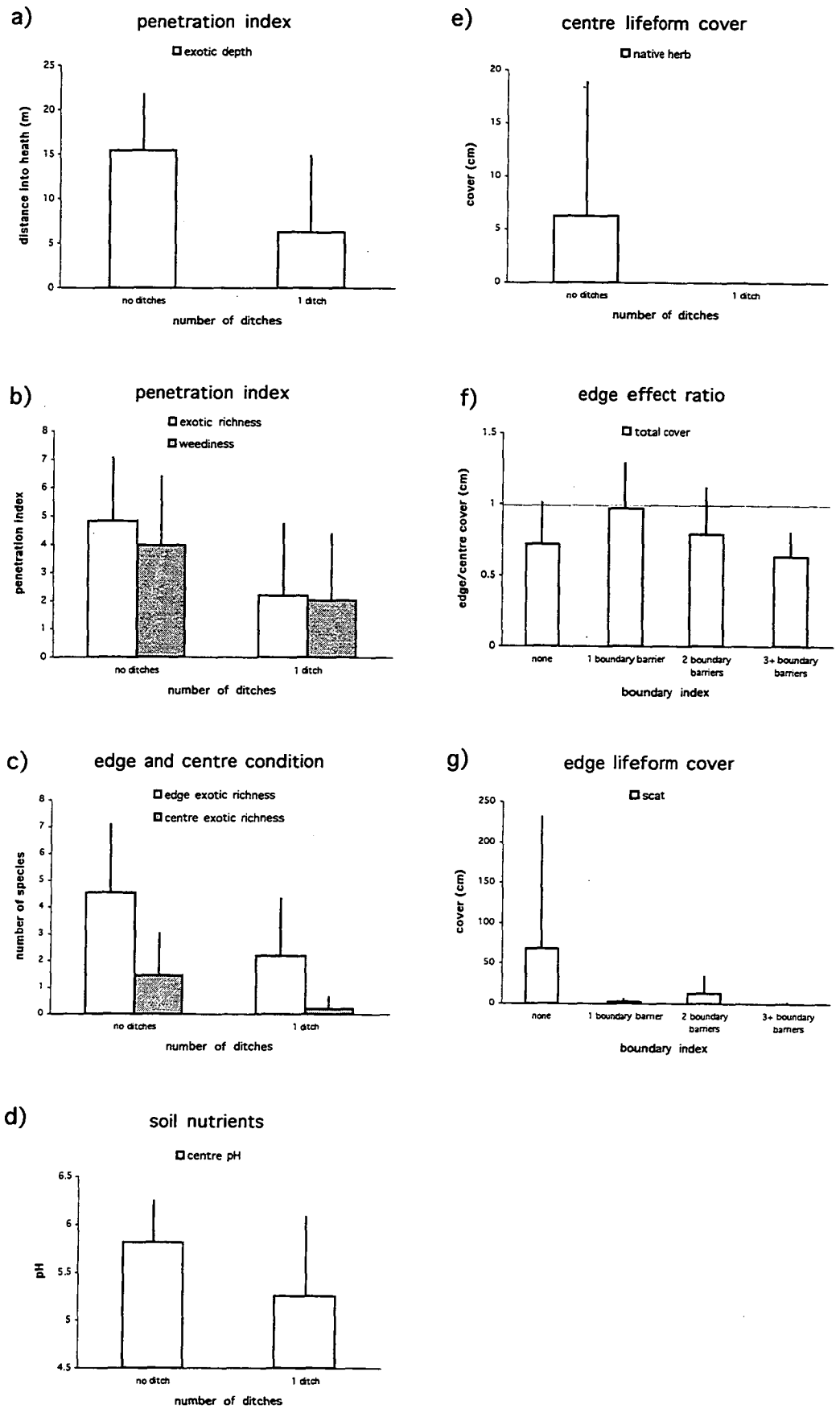


Figure 5.36. Differences in a–e) penetration indices, edge and centre condition, soil nutrients and centre lifeform cover for the number of ditches at the heath–pasture boundary, and f–g) edge effect ratio and edge lifeform cover for the total number of different barriers at the heath–pasture boundary. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness and penetration indices

Total cover was also affected by fences: with one fence there was roughly even total cover at edge and centre, whereas with two fences there was less cover at the edge than the centre (Figure 5.35h). This was most likely due to the more complicated boundary two fences produced, making it harder to define a sharp edge. There were fewer effects on native heath plants from barriers at the heath–pasture boundary. Native herb cover and herb % cover at the edge were lowest with two fences (Figure 5.35e). There was also less native herb cover when there was a ditch (Figure 5.36e). A track at the edge acted as a boundary to native grasses, seen as higher grass % cover if there was no track and lower if a track was present.

The boundary index, which was the sum of all the different barriers between the heath and the pasture (fences, tracks, ditches) had no significant effect on the exotic penetration indices. However, the ratio of total cover between the edge and the heath centre decreased as the boundary index increased, with the exception of no barrier (Figure 5.36f). As suggested above, more barriers (such old fencelines, overgrown ditches and a series of tracks) create a complicated and diffuse boundary with an edge that is more difficult to define. The condition variables at the edge and centre responded in a complex way that can be summarised as a decline in scat cover and exotic grass % at the edge and native grass % cover at the centre as the number of barriers increased (Figure 5.36g). In the Northeast region, there was a relationship between soil condition and weeds, with a slight increase in soil alkalinity and exotic richness at the edge for one type of boundary, but both being lower when there was none or two or more types of boundaries.

The disturbance index was the sum of all disturbance variables, i.e. a score of 1 meant there was none or only one type of disturbance, 2 meant there were two types, etc. The highest score of 4 included four or more types of disturbance (for edge quadrats) or, for centre quadrats, 3 included three or more types of disturbance. The highest disturbance score for centre quadrats is lower than for edge quadrats because there were generally less types of disturbance in the heath interior. Most quadrats had a value of 2. Although there was a disturbance score for all quadrats, only the edge and centre disturbance values are reported.

Disturbance at the heath edge had a highly significant effect on the presence of exotics. As the number of different disturbances at the edge increased, the cover and richness of exotic species increased relative to the centre, with the exception that the lowest disturbance index value had a high edge/centre ratio. The weediness index at the edge also increased (Figure 5.37a–c). A similar pattern was found in the Northeast region for exotic richness % and exotic grass cover (increasing with increasing disturbance). In other words, generally weed invasion increased as the number of disturbances at the edge increased, however in some cases weed presence was high even without many types of disturbance. It could be that the one type of disturbance actually had a bigger impact than

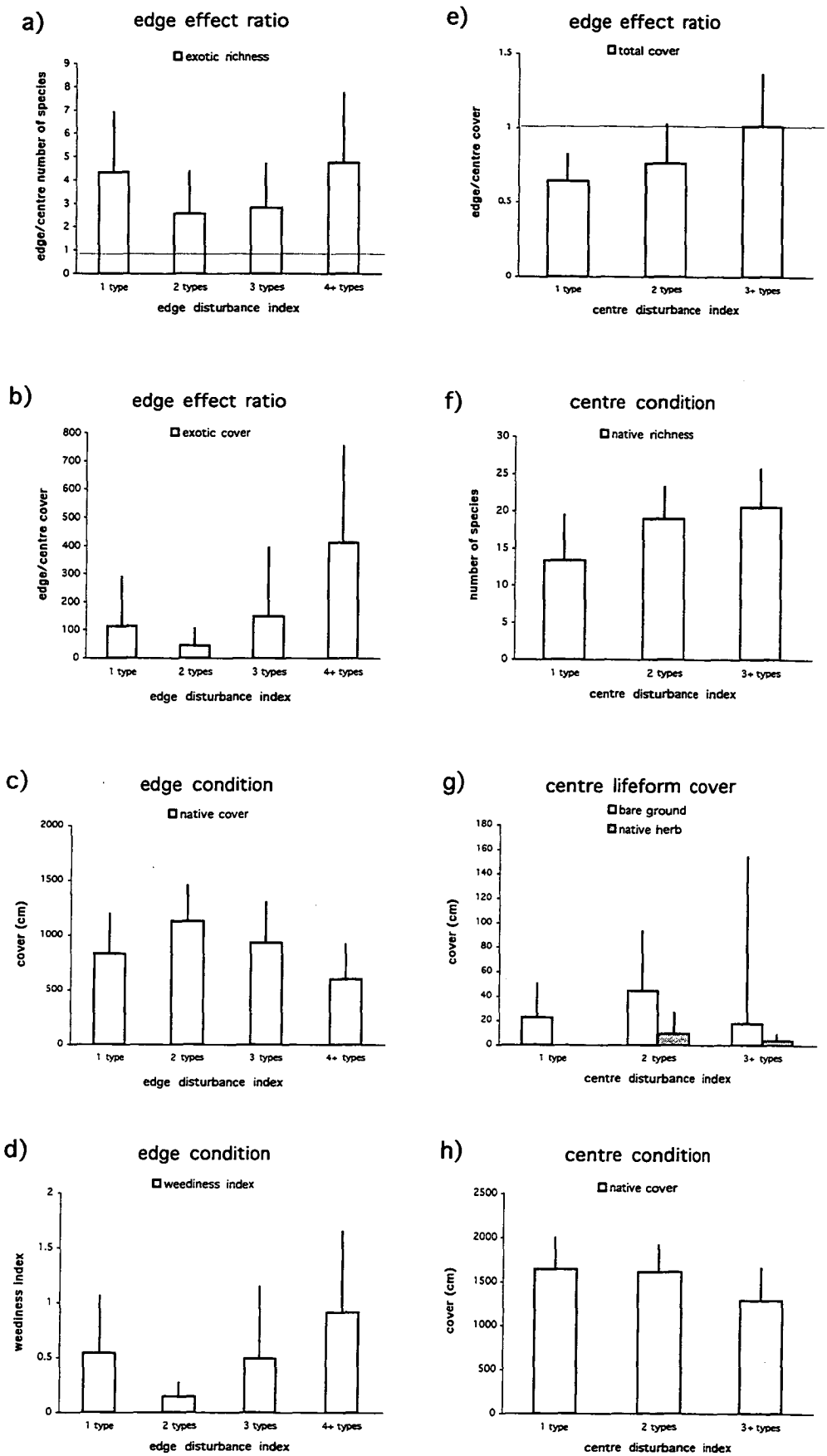


Figure 5.37. Differences in a–d) edge effect ratios and edge condition for the disturbance index at the heath edge, and e–h) edge effect ratio, centre condition and centre lifeform cover for the disturbance index in the heath centre. Horizontal line indicates equal amount of cover or species richness between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness index

a combination of other types. Disturbance at the edge affected native cover in the opposite way, that is, it tended to decrease as the number of types of disturbance increased (Figure 5.37d). In the Northeast region, the edge/centre native cover ratio, native cover at the edge and native shrub cover decreased as disturbance increased. In other words, fewer types of disturbance at the edge were generally associated with relatively more cover and a higher species richness at the edge.

In the heath centre, the disturbance index only had a significant effect on native variables, not on exotics. The total cover edge effect ratio, native richness, native herb cover and bare ground generally increased with more types of disturbance, but native cover decreased (Figure 5.37e–h). In the Northeast region, native herb % cover peaked for a score of 2 and, in the Other region, native richness actually increased as the number of types of disturbance increased.

The above results generally show an association between increasing number of different disturbances and increasing exotic cover and species richness. There is also an association between increasing number of different disturbances and decreasing total or native cover. Confirmation of this result was sought by testing the ratio of edge/middle disturbance index scores versus edge/middle species richness and cover. The results were that, as edge disturbances increased, the edge total species richness increased. There was no relationship when native and exotic species richness were tested separately. The opposite occurred with cover, as the number of edge disturbance got higher, the edge cover got lower. When total cover is split into native and exotic cover, there was a different disturbance effect for each type. As edge disturbance increased native cover at the edge decreased, but exotic cover increased. These results provide a clear summary of the effect of disturbance: species richness increases generally, but native cover declines and exotic cover increases.

When each disturbance was tested separately, tracks, trampling and slashing had no significant effects on the dataset as a whole. However, the presence of tracks was associated with higher native grass cover in the heath centre in the Other region. The formation of tracks may have increased the nutrient levels or provided an open environment that suited the grasses. Native herbivore grazing had a positive effect on native plant growth, with native grass cover at the edge and centre and native richness at the centre high if native grazing was present (Figure 5.38a, b). For the Other region, native cover was also high if native herbivore grazing was high. For animal digging, exotic cover and grass cover at the edge and exotic herb cover at the centre were higher, and exotic grass cover at the edge in the Northeast region was higher when diggings were present (Figure 5.38c–e). It is also possible that both diggings and exotic species were responding to another factor, such as more fertile soils.

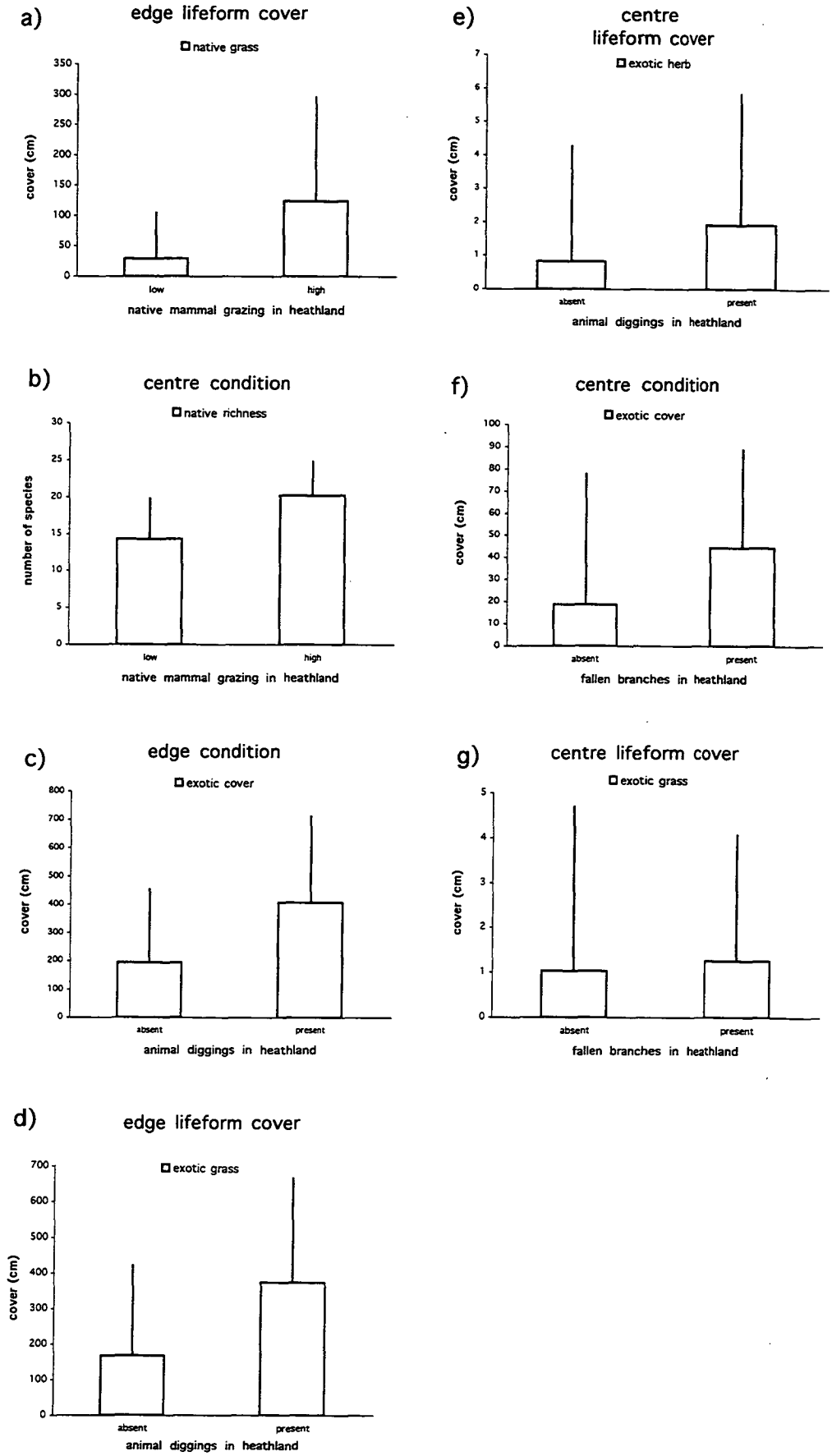


Figure 5.38. Differences in a–b) edge lifeform cover and centre condition for level of native mammal grazing in the heathland, c–e) edge condition and edge and centre lifeform cover for presence or absence of animal diggings in the heathland, and f–g) centre condition and lifeform cover for presence or absence of fallen branches in the heathland. Error bars show 1 standard deviation.

Other physical disturbances that affected the heathland were fallen branches, wind pruning and soil disturbance. When fallen branches were present in the centre of the heathland, there was a corresponding increase in exotic cover and exotic herb cover (Figure 5.38f, g). Perhaps the fallen branches provided safe sites or soil disturbance that was conducive to exotic seedling establishment. Wind pruning only affected the heathland edge, and was associated with higher exotic cover, exotic grass cover and weediness index but lower native richness, native cover and exotic richness % (Figure 5.39a–d). These results are probably biased by region, as all sites with wind pruning where found in the Northeast region. However, when the Northeast was tested separately, the same pattern was seen, native cover was lower and exotic/native cover and weediness were higher if there was obvious wind pruning. One would expect that native vegetation would be more able to withstand wind pruning than exotics, but perhaps the wind pruning occurred in conjunction with another disturbance that tipped the balance in favour of weed invasion. Soil disturbance at the edge resulted in lower native richness (Figure 5.39e). Rubbish combined with clearing at the edge was associated with less native graminoid cover, and animal trampling combined with soil disturbance in the centre was associated with lower native cover (Figure 5.39f, g).

The last type of disturbance that affected the indices of penetration was the presence or absence of *Phytophthora cinnamomi* symptoms. The presence of *P. cinnamomi* symptoms were associated with a lower weed presence overall. When symptoms were absent at both edge and centre, weed penetration was deeper; when symptoms were absent at the edge there was higher exotic richness and weediness; when symptoms were absent at the centre, there was a higher number of exotics penetrating the heath and higher centre exotic richness, cover, herb cover and weediness (Figures 5.40a–c, 5.41a–e). A similar pattern was observed for the Northeast region (edge exotic richness, exotic richness %, exotic/native richness higher if there was no sign of *P. cinnamomi*). Absence of pathogen symptoms at the heath edge was associated with slightly lower native cover at the edge rather than the centre (Figure 5.40d). Native richness in the Other region was also lower at the edge compared with the centre for absence of *P. cinnamomi* symptoms. The soils were more alkaline when there was no sign of *P. cinnamomi* (Figures 5.40e, 5.41g).

DISCUSSION: MANAGEMENT OF HEATHLAND REMNANTS

The results of this survey generally support the idea that the major disturbance that encourages weed invasion in heathlands is an increase in soil fertility. This was also obvious in the effect that an increase in the pH of the soil had on heathland condition. Environmental variables that showed an affinity with increased weed invasion had, in most cases, relationships with soil variables. This relationship between increased soil

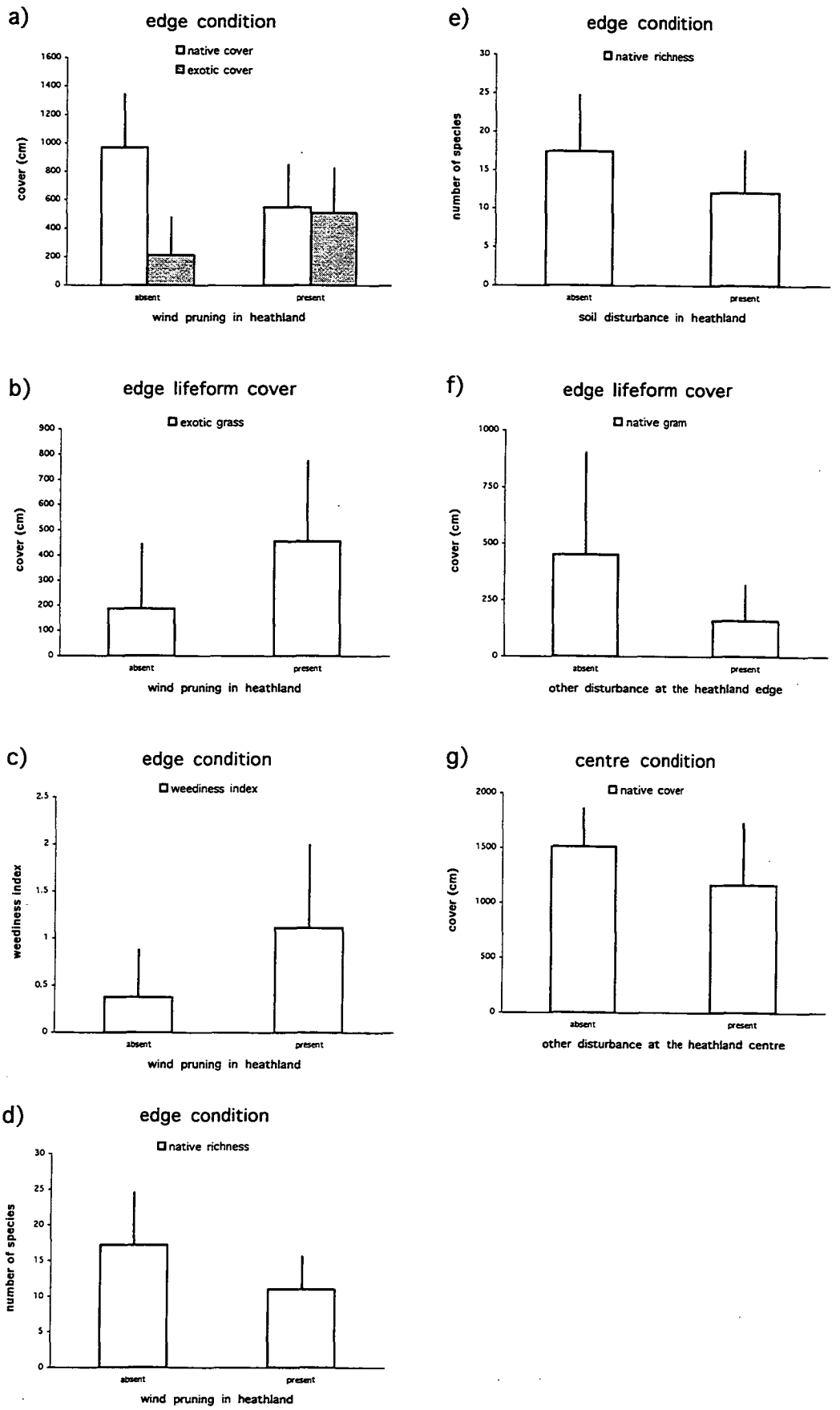


Figure 5.39. Differences in a–d) edge condition and lifeform cover for presence or absence of wind pruning in the heathland, e) edge condition for presence or absence of soil disturbance in the heathland, and f–g) edge lifeform cover and centre condition for presence or absence of other disturbance in the heathland edge or centre. 'Gram', graminoid. Error bars show 1 standard deviation. See text for description of weediness index and 'other disturbance'

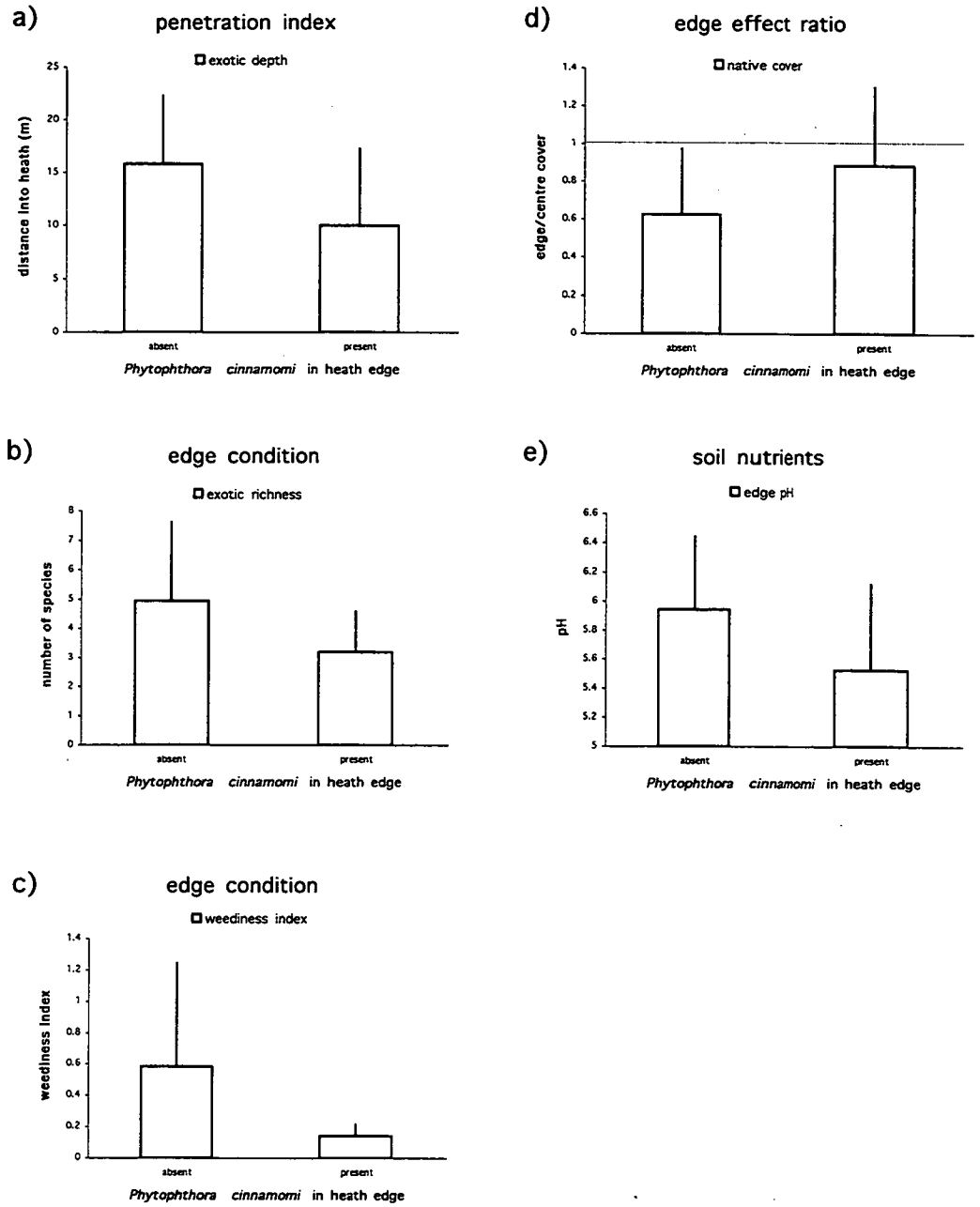


Figure 5.40. Differences in a) penetration index, b–c) edge condition, d) edge effect ratio, and e) soil nutrients for presence or absence of *Phytophthora cinnamomi* at the edge of the heathland. Horizontal line indicates equal amount of cover between heath edge and centre. Error bars show 1 standard deviation. See text for description of weediness and penetration indices

nutrients and higher weed invasion at the boundary between native vegetation and agricultural land has been seen by many other researchers, for example Grigg *et al.* (2000).

Another major finding from the survey is that the edge effects did not penetrate very far or cause a major loss of condition in the heath. Although exotic species were found in great numbers at the edge and were occasionally found further than 20 m into the heathland, Their numbers and cover dropped sharply after 5 m. Thus the transect distance was adequate to sample the presence of most exotic species. Hester and Hobbs (1992)

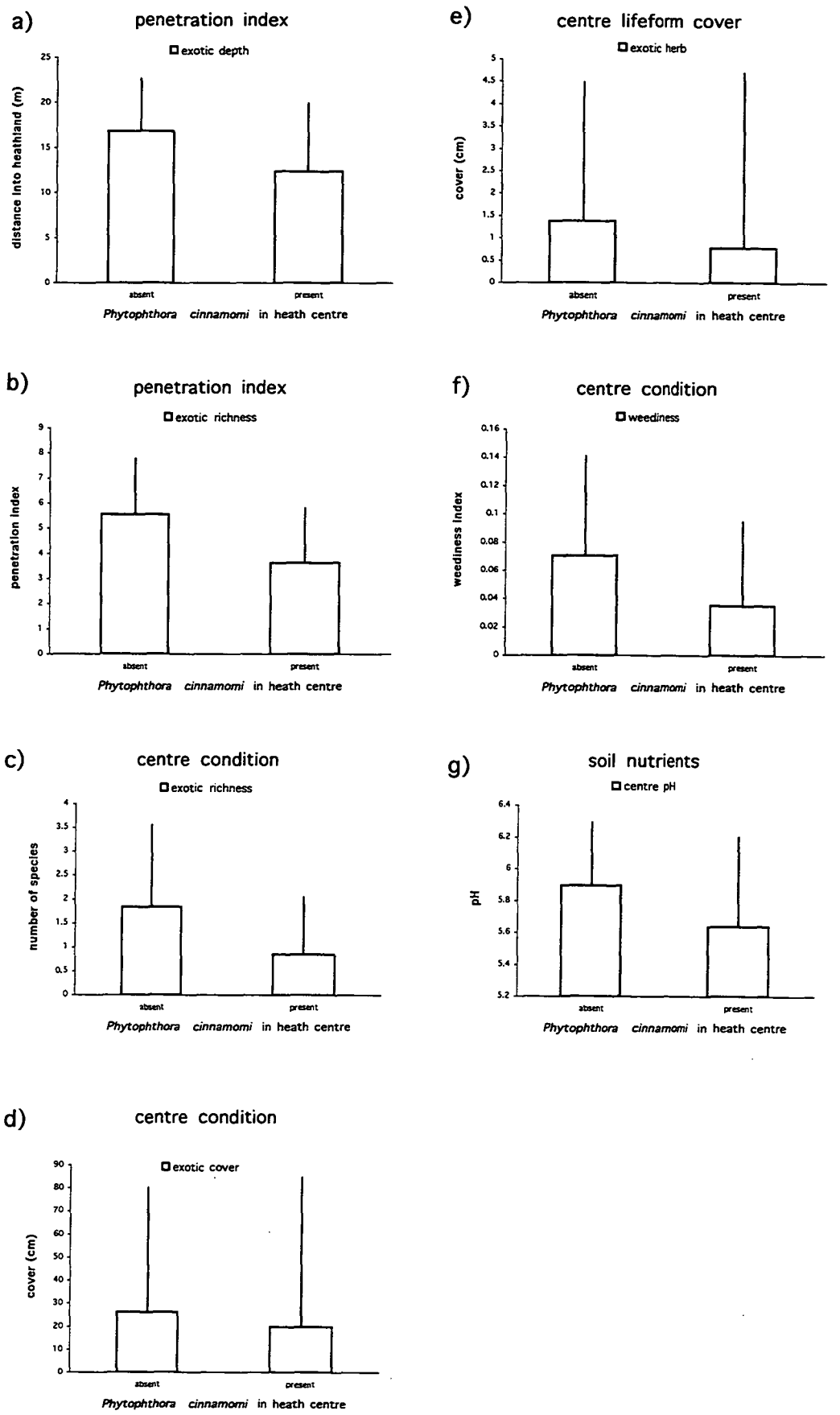


Figure 5.41. Differences in a–b) penetration indices, c–f) centre condition and lifeform cover and g) soil nutrients for presence or absence of *Phytophthora cinnamomi* in the heathland centre. Error bars show 1 standard deviation. See text for description of weediness and penetration indices

also found a high abundance of exotic species at the edges of shrubland vegetation remnants, but they similarly declined dramatically just a few metres from the edge. Dry sclerophyll vegetation generally, and heathlands in particular, appear to be quite resistant to weed invasion from their boundaries (Amor and Stevens 1975; Hobbs and Atkins 1988; Hester and Hobbs 1992).

In contrast to the evidence provided by other research into habitat fragmentation, the current study did not find any great effect of the age of the heathland boundary. There was a minor decrease in exotic grass and native herb cover for older boundaries that could be related to grazing pressure or soil nutrients. Other research has found an increase in extinction rates for the number of years chaparral vegetation was isolated in fragments in canyons (Soulé *et al.* 1992) and increasing invasion by woody weeds with increasing age of urban bushland sites (Rose 1997).

Soil nutrients and fertilisation

Phosphorus

A gradient of soil nutrients over a remnant boundary such as was evident in this survey has also been found by Muir (1979), Cale and Hobbs (1991), Hester and Hobbs (1992), Scougall *et al.* (1993) and Grigg *et al.* (2000). Phosphorus was positively correlated with most of the exotic cover, richness and lifeform variables, particularly at the heath boundary. There were many relationships between phosphorus levels and the penetration of exotic species into the heathland. Where higher phosphorus levels were measured for flatter slopes, flat landforms, and heath adjacent to pasture on reserved land, exotic variables were also found to be high. These patterns are only speculated to be causative, but there is a consistent broad picture of higher nutrient levels being associated with increased weed invasion, supporting the findings of many other researchers for heathlands (Specht 1963; Connor and Wilson 1967; Heddle and Specht 1975) and for other vegetation types in Australia (Hobbs and Atkins 1988; Cale and Hobbs 1991; Hester and Hobbs 1992).

In general, when there was a high phosphorus/nitrogen ratio at the heath edge there was more plant cover at the heath edge. In some sites this was obvious in the field as a small but distinct 'hedge' of plant growth, alike to that found by Grigg *et al.* (2000) at the edge of a *Banksia* woodland. Much of the research into the effects of nutrient additions on heathland has shown more dramatic increases in native shrub growth or cover (Specht 1963; Clark 1975; Heddle and Specht 1975; Specht *et al.* 1977). However, such research has involved much higher levels of added nutrients. In some instances in the present study, such as for the presence of cattle, for recent pasture fertilising and for heathland on crown land, high phosphorus levels and high native species richness occurred together. This is

perhaps surprising given the observed negative response heath seedlings have to 'phosphorus toxicity' (Specht 1963; Heddle and Specht 1975; Specht *et al.* 1977). Again, the response found in the present study is probably explained by the relatively low levels of nutrient drift from agricultural land. However, not all responses by native vegetation to phosphorus were positive. Native richness was relatively low for moderate wallaby grazing, native grass cover was low for pasture that was on reserved land and native cover (particularly herb cover) was low for flatter slopes, all coinciding with high phosphorus levels. There are many examples from other heathland research where growth has been reduced or there was no response at all when phosphorus was added (Specht 1963; Grundon 1972; Heddle and Specht 1975; Specht 1975; Groves and Keraitis 1976; Specht *et al.* 1977; Ozanne and Specht 1981). The individual plant responses to the addition of nutrients would have to be investigated at the species level to tease out the real relationships between phosphorus and changes in native cover.

Nitrogen

Nitrogen levels in the soil were also associated with the presence of exotics and changes in growth patterns. There was a simple positive correlation between the amount of nitrogen at the heath edge and weed penetration. There were many other associations between higher nitrogen levels, grazing and increased weed invasion, such as for less than 6 months per year of pasture grazing, when cattle were present in the heath, and when Cape Barren Geese were present in the pasture. Flat landforms also had high native grass cover and higher nitrogen levels. Again, these results do not show causation, but they do show a common pattern which supports the findings of other researchers, that nutrient additions can increase weed invasion in nutrient-poor heathland ecosystems (Specht 1963; Heddle and Specht 1975; Amor and Piggin 1977; Heil and Diemont 1983; Aerts and Berendse 1988; Aerts *et al.* 1990; Mendoza *et al.* 1995; Posse and Mendoza 1995). An interesting finding of the present survey is that soil nitrogen levels did not significantly relate to any of the disturbance variables.

The increased growth response of native plants to nitrogen measured by other researchers (Connor and Wilson 1967; Specht *et al.* 1977; McMaster *et al.* 1982; Witkowski *et al.* 1990b; Angold 1997) was likewise found in the present study. Relatively high nitrogen levels were associated with high native cover for pasture fertilisations that were close together; high native grass cover for flat landforms and if pasture grazing occurred all year; high fern cover in the northeast for current grazing in pasture; high graminoid cover for darker soils and high stocking rate in the pasture; and high native shrub cover for crown land and if the pasture was sown more than once. As nitrogen was most usually supplied by sowing clovers and subclovers, any increase in nitrogen availability will extend at the same pace as the spread of exotic legumes. This can be illustrated by the association between the use

of a super spreader (which can be used to apply clover seed in conjunction with super phosphate), higher nitrogen levels and higher native graminoid and native herb cover at the heathland edge. Individual species can react quite differently depending on the amount of nitrogen and its ratio with phosphorus (McMaster *et al.* 1982; Witkowski *et al.* 1990a; Witkowski *et al.* 1990b). There was a correlation between high nitrogen and low native species for when the pasture was grazed for more than 6 months, when cattle were present in the heath, and when a super spreader was used for sowing seed. This seems to indicate a limitation of growth for some native species with higher concentrations of nitrogen as found by Heil and Diemont (1983).

Soil pH

The acidity or alkalinity of the soil played a very important role in weed invasion and the condition of the heathlands in this survey. The various associations between pH and soil hue, liming and the weed penetration indices all point to increased weed invasion when the soils were less acidic. This took the form of higher exotic cover (particularly for herbs) and increased exotic species richness. These results are logical given that pasture species prefer more neutral conditions. There was also a general pattern between more acidic soils and lower native grass cover. There were many instances where higher soil nitrogen levels were found in conjunction with more acidic soil, the reverse of what usually occurs in mineral soils, where nitrogen becomes most available at a pH of about 6.5, due to greater rates of microbial nitrogen mineralisation (Pywell *et al.* 1994; Miller and Donahue 1995).

One simple way of managing nutrient increases and weed invasion would be to leave a buffer zone between the heathland edge and agricultural areas. This was suggested by Specht and Cleland (1961) as early as 1961, where they proposes a buffer zone of 400–800 m around nature reserves containing dry sclerophyll forest, sclerophyllous mallee and heath formations. Weed invasion in the present survey dropped markedly before 20 m into the heath, so even a small buffer zone might be quite effective. This could be achieved by not applying phosphorus fertilisers or lime or not sowing exotic legumes right up to the heathland boundary.

Grazing in pasture and heathland

Weed invasion

In the field there was ample visual evidence of stock use of heathland for grazing and shelter (trampled ground, broken plants, faeces) where heathland–pasture boundaries were unfenced. Williams (1991) discusses the obvious impacts of grazing in dry sclerophyll

vegetation in Tasmania as including the effects of ground compaction and trampling, puddling on wetter sites, introduction of exotic species and loss of recruitment to sapling and tree layers. To some extent, the visual evidence of adverse stock impacts in the present study was verified by statistical support for an association between stock grazing and weed invasion, as has been found in other studies (Gilfedder and Kirkpatrick 1993; Scougall *et al.* 1993; Pettit *et al.* 1995). There was a relative increase of weeds in the heath centre compared with the edges with the presence of cattle in the pasture (Figure 5.17b, c). The fact that cattle had more of an impact than sheep might be due to the different types of country grazed by domestic stock in Tasmania. Cattle are generally found in areas of better rainfall, which also provide better growing conditions for exotic species. Cattle can affect the edges of other vegetation types, such as temperate rainforest, by trampling the understorey, allowing light to penetrate and acting as vectors for seeds of exotic coloniser species (Fox *et al.* 1997). However other stock (such as pigs, camels and horses) appeared to be relatively benign, although there were not enough transects to be able to provide much detail about their individual impacts.

The association between grazing and weed invasion could sometimes be explained by changes in soil nutrient levels. For example, the deepest weed penetration and the highest nitrogen levels were both found when the pasture had been grazed for a short period of time (1–5 years). Scougall *et al.* (1993) also found higher nitrate-nitrogen and ammonium-nitrogen levels in open woodland remnants accessible to domestic stock. The observations in the present study do not indicate a direct causal relationships but again show a logical pattern, one that is supported by other researches for other vegetation types (Hobbs 1991; Pettit *et al.* 1995; Fox *et al.* 1997; Yates *et al.* 2000). Another series of associations, between increased exotics, higher soil phosphorus, higher soil nitrogen and the presence of Cape Barren Geese, verifies the observation that Geese prefer the good feed provided by well-fertilised pasture (Morrison and Purling 1982). There is also a relationship between grazing, weed invasion and the soil pH. In some cases where grazing was associated with decreased exotic richness, the soil was also more acidic, such as for 11–20 years of grazing. This is in contrast to the more acidic soil that occurred after domestic stock grazing as seen by Scougall *et al.* (1993). However, exotic species would be more likely to grow on more alkaline soil over the usually acidic heathland soils.

Native herbivore grazing also resulted in a general increase in weed invasion, although not always in a linear way. Moderate rather than heavy or severe wallaby and wombat grazing occurred in conjunction with the highest cover and deepest penetration of exotics. These mammal herbivores would have been getting their main meal from the pasture each night then returning to the heathland during the day for cover (Leigh and Holgate 1979). A clear mechanism was shown to be the trails that these mammals make through the heathland from the pasture. The bare ground along the trails assist in the establishment

of exotic species, allowing the native herbivores to act as vectors for weed invasion. Animal diggings also encouraged exotic invasion, particularly in the Northeast region. Diggings disturb the soil and so provide safe sites for seedling germination. This finding is supported by other research that has shown that soil disturbance alone can be enough to enhance weed invasion in heathlands (Clemens and Franklin 1980). The fact that weed invasion was more severe when native herbivore grazing was moderate could be explained by preferential grazing of exotics when grazing becomes more heavy or the combination of grazing by different herbivores or a lack of precision in the categories of 'light', 'moderate' and 'heavy' by the owners and managers asked about native grazing. However, in a like manner to domestic stock grazing, there is a relationship between high phosphorus levels, greater weed penetration and more weed cover at the edge. Other variables possibly connected are high exotic cover, low native cover and relatively alkaline soils.

There was a certain amount of variability in the results described in the present study, resulting in some exceptions to the general association between increased grazing, increased nutrients and increased weed penetration. One such example is where there was a delay between the presence of grazing and the presence of weeds. This occurred in the case where exotic herb cover was higher when the pasture was grazed 1–5 years ago compared with it being currently grazed; and when exotic herb cover was highest when grazing occurred more than 5 years ago compared with more recent grazing. Exotic richness was also lower when stock grazing was currently present in the heathland. These results could be explained by domestic stock grazing temporarily masking weed invasion by preferential grazing of exotic species when the stock are present. The stock may provide the right conditions for exotics to become established some years after grazing has ceased. This suggested mechanism is the opposite of that proposed by Amor and Piggin (1977), where stock selectively graze palatable native species and avoid unpalatable (prickly or thorny) exotic species. Another exception to the association between more grazing and more exotics is the low penetration index for exotic cover found when grazing occurred more frequently (in terms of months per year). One possible explanation is that the amount of grazing present and the amount of degradation of a native bush remnant may not be directly related, as was apparent with wallaby and wombat grazing.

Fencing or providing some other barrier at the heath–pasture boundary would reduce the amount of weed invasion into the heathland. The present study shows that more barriers at the boundary are associated with less weed penetration. All the fences seen during the survey were constructed in such a way as to limit domestic stock movement but not native grazers. Thus, any reduction in weed invasion found with fencing is likely to be due to a reduction in access by domestic stock. Fencing out domestic stock reduces the decline of soil structure and native species richness and cover, circumstances which would eventually encourage weed invasion (Scougall *et al.* 1993).

Changes to native cover

Grazing by domestic stock and native animals in the present study was associated with both decreases and increases in native plant cover. A decline in native cover and/or density due to grazing is common (Leigh and Holgate 1979; Dickinson and Kirkpatrick 1986; Gilfedder and Kirkpatrick 1993; Scougall *et al.* 1993; Pettit *et al.* 1995). The type of grazing associated with a decline in native cover at the heath boundary in the present study include moderate levels of feral animal grazing (rabbits and hares), and the presence of cattle in the pasture. Declines in cover in the centre were found in the presence of more regular grazing in the heathland. Dickinson and Kirkpatrick (1986) also measured reductions in native cover as a result of increased grazing pressure by native and feral herbivores. There is some evidence to suggest that the destruction of the canopy can make a significant difference to the invasibility of heathland communities. In a shrub-dominated community with a dense shrub cover in the WA wheatbelt, a relationship was observed between reduced shrub cover and increased weed invasion (Hobbs and Atkins 1991). Exotic species are generally more abundant at the edges of native vegetation remnants, but their penetration into the centre can be hindered by the closed edge of a heathland compared with a more open woodland canopy (Hobbs and Atkins 1988; Hester and Hobbs 1992). Conditions at the edge can therefore influence the invasibility of the centre, particularly the density of the shrub or overstorey layer (Hobbs 1991). One of the results of the present study suggest that this relationship could be in operation — with the presence of cattle seemed to come a decrease in cover at the heath edge and an increase in weed penetration into the heath centre. Fox *et al.* (1997) also found a relationship between trampling of the understorey by cattle and an increase in the proportion of colonisers at the edge of a temperate rainforest.

In certain situations, increased grazing actually appeared to increase native cover. This occurred for shrub cover when 'other stock' were present in the pasture and for current stock grazing in the heathland. Grazing by domestic stock in Australia generally results in an increase in woody species (Wilson 1990), and shrubs can increase in abundance if they are of low palatability and reproduce vegetatively (Wilson 1990). In other situations in the present study, moderate levels of grazing occurred in conjunction with high levels of native cover, such as for 11–20 years and 6–11 months per year of pasture grazing and moderate wombat grazing. When the grazing pressure increased the cover decreased. A possible mechanism may be the increasing acidity of the soil, seen in the present study for current stock grazing in the heathland and moderate wombat grazing. This is consistent with the evidence provided by Scougall *et al.* (1993), where soil in open woodland remnants became more acidic when the remnants were grazed by sheep and other domestic livestock. A relative increase in cover at the edge compared with the centre for the highest stocking

rate in the pasture was also seen in the present study. This more distinct boundary could be due to more intense grazing management and thus a better maintained pasture.

Changes to native species richness and composition

Grazing can change native vegetation structure and composition depending on the amount of grazing pressure, the different feeding preferences of the different herbivores, which plant species are palatable and their response to grazing (Leigh and Holgate 1979; Kirkpatrick and Harris 1999a). In the present study, native richness was lower when heathlands were currently grazed and when the pasture was grazed all year rather than part of the year. It was not clear whether these declines were due to selective grazing or increased disturbance and trampling or some other factor. Some decreases in native richness could be related to increases in the soil nutrients, for example, nitrogen levels at the heath edge were high when cattle or sheep were grazing in the pasture, and phosphorus levels were high when there was moderate wallaby grazing. Species richness may decline because some heathland species have reduced seedling establishment when phosphorus and nitrogen levels are high (Specht 1963). Other species are then able to compete more successfully, such as annual grasses and herbs (Specht 1963).

In opposition to this theory were the results where native richness was higher for certain types of grazing and where the soil nutrients were also higher. This occurred when other stock were present (higher nitrogen), when cattle were present in the pasture in the Other region (higher phosphorus and nitrogen and more acidic soil), and when wallaby grazing was high in the Other region (higher phosphorus). It is difficult to know what mechanism was in operation without knowing exactly which species increased. There is evidence to suggest that, for environments with a long history of grazing, some amount of herbivore grazing is necessary to maintain species richness and community condition (Kruger *et al.* 1989; Pettit *et al.* 1995). Coastal heathlands are unlikely to be adapted to grazing by domestic stock but could be described as having a long relationship with native herbivores.

Some particular lifeform groups benefited more than others from the disturbance caused by grazing. Fern cover — probably mostly *Pteridium esculentum* (Austral Bracken) — and graminoid cover were both high for different grazing circumstances. Bracken is well able to withstand disturbance and is unpalatable to stock, as are the silica-rich graminoids. Domestic stock would probably prefer softer herbaceous species such as those found in Asteraceae, Poaceae, Liliaceae and Orchidaceae, and also regrowth (particularly after fire) of some shrub species (Kirkpatrick and Harris 1999b). In the present study, native herb cover was lower when grazing occurred all year and when sheep were present in the pasture. Herbs are usually the most affected by grazing, due to their relatively high palatability (Wilson 1990).

Sowing

Agriculture provides the ideal opportunity for exotic invasion into native vegetation: the provision of exotic seeds and disturbance through mechanical spreading of the seed. Two pasture species were associated with increased exotic herb cover, grass cover or species richness, and these were *Dactylis glomerata* (Cocksfoot) and *Festuca* species (Fescue). Due to the difficulty of identifying grasses to species level in the field, it is not clear whether Cocksfoot or Fescue were present in heathland. However, one of the exotic grasses that was identified in the heathland was *Festuca arundinacea* (Tall Fescue), a common pasture grass in Tasmania. *F. arundinacea* is widely adapted to many soil types and is particularly well suited to situations of waterlogging or mild salinity, and also responds well to drought (Lane *et al.* 1999). In other words, it is well adapted to the soil conditions usually found in heathland. Exotic grasses can be particularly invasive if there is an increase in nutrients and/or disturbance, as found in a dry heathland in Tierra del Fuego (Mendoza *et al.* 1995). Other ways of providing seed, such as roadsides and cars, have also been shown to increase weed invasion (Amor and Stevens 1975; Wace 1977; Rose and Webb 1994). However, this suggested invasiveness of Fescue is only speculation, and it is also quite likely that it is only a marker for a set of environmental or management conditions that encourage weed invasion generally. Regional differences are probably also at least partly or possibly mostly responsible.

The few positive responses of native cover to the sowing variables show that sowing was not a major factor influencing heathland condition. More regular sowing and a longer period of time since the last sowing meant a more distinct boundary and more native grass cover, respectively. These responses could be due to the disturbance of actually applying the seed and/or a fertiliser effect rather than the exotic seeds themselves (Hobbs 1989).

Heath management and other disturbance

Weed invasion

Disturbance has long been associated with weed invasion (Fox and Fox 1986; Hobbs 1989; Rejmánek 1989), and the results from the present survey clearly show this response. When the disturbance variables measured (tracks, grazing, animal diggings, broken or fallen vegetation, slashing, wind pruning, rubbish dumping and soil disturbance) were combined into a disturbance index, the higher the score the higher the weed invasion. These disturbances change the soil structure, add nutrients, open the canopy and provide patches of disturbed bare ground, all ideal conditions for weed invasion. In some cases weed invasion was high despite a low disturbance index score. This can be explained by the difficulty of creating the disturbance index. A higher score was due to more different types

of disturbance, but all types of disturbance were not equal and some (such as domestic stock grazing) may have had more impact than others (such as a low-traffic track).

The individual disturbance variables that encouraged higher weed cover and richness were slashing and clearing, fallen branches and wind pruning. The establishment and growth of invasive exotic species depends on there being suitable 'safe sites' consisting of bare ground where the competition of native species is temporarily removed (Lepart and Debussche 1991). Slashing and clearing provide such sites and, in the present study, were associated with less exotics at the edge but more exotics penetrating the heathland centre. Fallen branches also seemed to encourage exotics and might similarly have created soil disturbance and areas of bare ground. Coastal heathlands are often subject to wind pruning, particularly by salt-laden winds, causing asymmetrical growth (Kirkpatrick and Harris 1999b). In the present study, such wind pruning was found in conjunction with comparatively high exotic cover at the heathland edge. One explanation is that the reduction in native cover (that was found when wind pruning occurred) allowed more exotics to establish. Another explanation is that nutrients provided in the sea spray aided exotic establishment.

Not all types of disturbance measured in the present survey resulted in weed invasion. Time since fire was not associated with weed invasion, and this supports the findings of other researchers into pyric succession in heathland (Hobbs and Atkins 1991; Hester and Hobbs 1992). Larger shrubs have often been found to increase in abundance, cover levels or density over time since fire (Keeley *et al.* 1981; Posamentier *et al.* 1981; Morrison *et al.* 1995), although this is dependent on length of inter-fire interval and the mixture of plants that regenerate by seed and that regenerate by resprouting.

The land tenure of both the heathland and the pasture had a complex relationship with the presence of exotics. The way private land was managed appeared to maintain heathland condition, with generally low levels of exotics at the heath edge. In comparison there were more exotic species found at the edges of reserved heathland. This result is in contrast to other studies in Tasmania showing a reduction in remnant condition on private land (Gilfedder and Kirkpatrick 1998). The nutrient levels in the soil may again have some part to play in this result, as phosphorus at the edge was higher for crown land and other reserves. Region may have also contributed, as most of the 'other reserve' sites were in the Waterhouse Conservation Area (a type of reserve having less restrictions than a national park) which was adjacent to privately owned pasture.

In the field, land tenure of either pasture or heath did not necessarily translate into a particular land use of the heath. For example, some pasture that was national park had in the past been cleared and fertilised (e.g. Mt William National Park and Asbestos Ranges National Park), whereas some pasture that was freehold or leasehold was fertilised,

sown and grazed a long time before and consequently quite run down (e.g. The Sanctuary on the east coast of Flinders Island). Reservation status for the cleared area did not necessarily imply less disturbance into the heath, as sometimes the area was grazed due to neglected or absent fencing. It was hoped initially that land tenure might provide another way of summarising the disturbance level of the heath but, although there were relationships between tenure and weed invasion, there was more to the disturbance than just the land tenure. Another difficulty in using land tenure as an indication of disturbance level is that the boundary is affected by both the pasture and the heathland, which may have different land tenures.

The spread of fungus *Phytophthora cinnamomi* does not necessarily lead to invasion by exotic grasses and herbs (Weste and Law 1973), but it can cause dramatic changes in floristic composition. Severe dieback of susceptible species can occur, leading to an increase in less susceptible sedges, rushes and *Leptospermum* (Teatree) species (Weste and Kennedy 1997). In the present survey, the presence of *P. cinnamomi* symptoms were associated with lower weed invasion at both the edge and the centre of the heathland. There are three possible reasons for this finding. Firstly, that the conditions which are most conducive to the spread of *P. cinnamomi* are not conducive to the spread of exotic herbs and grasses. The conditions that favour the fungus are acidic shallow soils with poor drainage and warm wet periods followed by periods of water stress (Weste 1974), none of which are particularly good conditions for exotic herbs and grasses. Secondly, given the prevalence of the pathogen in Tasmanian heathlands (Kirkpatrick and Harris 1999b), it is possible that a score of *P. cinnamomi* absence actually indicated that the pathogen had passed through the heathland prior to this survey, allowing ample time for weed invasion. In which case, when *P. cinnamomi* was scored as present, it may mean that weed species have not had time to invade. The full effects of dieback and subsequent increase of grasses and exotics can take from 6–8 months to 3 years from the invasion of the pathogen (Weste and Law 1973; Weste 1974). Thirdly, *P. cinnamomi* may be present but with no obvious symptoms; this can occur when the fungus is found on relatively fertile soils. Of these three possible explanations for the results found in the present survey, the first is more likely, particularly when taking into account the more alkaline soil that was measured when *P. cinnamomi* was absent.

Changes to native cover

Two of the independent variables measured in the present survey were associated with an increase in native plant cover. The first of these variables was fire — the relative increase in cover in the heath centre is in accordance with other studies showing increasing shrub cover or abundance over time since fire (Keeley *et al.* 1981; Posamentier *et al.* 1981; Morrison *et al.* 1995). The second variable is land tenure, and privately owned heathland

showed increased cover of native plants at the boundary with pasture. There is a possible relationship between this high native cover and the lack of exotics at the edge of privately owned heathland, similar to that seen by Hobbs and Atkins (1991) between reduced shrub cover and weed invasion in remnants in the WA wheatbelt. Other similar examples from the present survey were observed between increasing types of physical disturbance, decreasing total or native cover and increasing exotic cover and richness.

In contrast to the positive responses just discussed, native cover generally responded negatively to disturbance, in particular, to clearing, animal trampling combined with soil disturbance and to a combination of different types of physical disturbance. These results are supported by other results showing that heathland vegetation is easily damaged, recovers slowly from physical disturbance, and experiences a loss in biomass (Clemens and Franklin 1980; Soulé *et al.* 1992; Rose and Webb 1994; Pelton and Conran 2002).

The boundary between heathland and pasture was in some cases difficult to define, consisting of a very gradual decrease in native cover and increase in exotic cover over distance. Two fences and other barriers (e.g. ditches and tracks) were found to create such complex boundaries. For the purposes of the present survey, this complexity had to be simplified, a problem that has dogged other research into edge effects (Murcia 1995). Although the pasture quadrats were not analysed in detail, the fact that there were such diffuse boundaries shows that some heathland plants (mostly shrubs and graminoids) were able to reinvade the pasture. This is another indication that heathland remnants can be relatively resistant to weed invasion and worth conserving.

Changes to native species richness and composition

The response of native species richness to disturbance depended on where the disturbance occurred. Increases in native species richness were generally found in the heath centre, in association with private land, clearing and physical disturbance. Possible causes are changes to the microclimate allowing ephemeral or light-demanding species to establish, lower phosphorus levels enhancing the usual high species richness found in heathlands, or the remnant size affecting the number of species. Remnant size was not measured in the present survey due to the focus on the boundary rather than the area. Other researchers have found that the richness of heathland remnants were negatively correlated with the size of the remnant (Bond *et al.* 1988; Webb and Vermaat 1990). In the present survey, declines in species richness were generally found at the boundary between heathland and cleared land, in association with clearing along the boundary and soil disturbance at the edge. Possible causes could be changes to the microclimate that were detrimental to heathland species and/or the increased competition from exotic species.

The native lifeforms that responded to physical disturbance all responded in the centre to the presence or absence of barriers at the heathland edge. Native grass and native herb cover was lower the more barriers there were at the edge, which was the same result as for exotics. Although there was no observed significant difference in soil nutrients, these results could still be related to other aspects of soil fertility.

CONCLUSIONS

Coastal heathlands were found to be relatively resistant, with the species numbers and cover values of exotics declining rapidly within a few metres of the heath–pasture boundary. The age of the boundary also did not have a major impact on heathland condition. These two findings mean that remnant patches of heath, whether well established or newly created, are unlikely to be overwhelmed by edge effects and are worth conservation effort. However, there are some management practises that would aid better heathland remnant conservation. Increased levels of phosphorus and nitrogen (and increased soil alkalinity) were implicated in weed invasion into the heathland. Increased nitrogen levels appeared to have a negative impact on native species richness. Care should be taken with the fertiliser application method close to the boundary, perhaps even leaving a buffer zone between the application of superphosphate or the sowing of pasture legumes and the heathland edge. Stock grazing, particularly of cattle, was generally associated with higher exotic presence and, in some cases, lower native grass cover. The two possible mechanisms were a relationship between grazing, weed invasion and increased nutrients levels, or a decrease in the protective dense layer of native shrubs. In some instances, previous domestic stock grazing may have masked a weed invasion problem that only appeared approximately 5 years later. Native mammals can act as vectors to bring exotic species into the heathland via their hard bare trails and soil diggings. Fencing or other barriers at the heath–pasture boundary were shown to aid in reducing weed invasion. The sowing regime in the pasture had no major negative impact on heathland condition. However, the role of *Festuca* species needs further study. In general, increased physical disturbance seemed to cause increased weed invasion and lower native cover. Native species richness at the heath edge suffered due to disturbance, but benefited in the centre. Weed invasion could be reduced by minimising the amount of tracks into the heath, not cutting wood for firewood, not dumping rubbish and not cutting or slashing the shrubs.

6. Discussion: Conservation of Heathland in Tasmania

This chapter makes the assessment of whether to accept or reject the null hypotheses outlined in the Introduction. The results from each of the field experiments and the heath–pasture boundaries survey are compared with each other and then used in making this assessment. The null hypotheses are that: 1) realistic applications of nutrients to heathlands have no effect on their species richness, composition and growth; 2) realistic applications of nutrients to heathlands, in interaction with fire, have no effect on their species richness, composition and growth; and 3) the nature of the disturbance regime on both sides of the heath–pasture boundary has no effects on heathland species richness, composition and growth. This chapter deals with each of these in turn and then discusses the implications for heathland conservation.

THE EFFECTS OF THE ADDITION OF SMALL AMOUNTS OF NUTRIENTS

The amounts of nutrients added experimentally in the present study were more realistic in terms of what is likely to occur as a result of human activities adjacent to heathlands, than the amounts used in experiments conducted by other researchers. In the present study, the amount of phosphorus used was approximately 2 and 4 kg/ha added in fertilisers; 2.4 kg/ha added in sewage wastewater; and less than 1 kg/ha added in fire-fighting foam. These are very small amounts when compared with those used in other experiments in heathlands, which range from approximately 13 kg/ha (Clark 1975) to approximately 170 kg/ha (Specht 1963).

Effects on growth and flowering

In general, there were no dramatic detrimental effects from the addition of nutrients in the present study. The lack of death and disease in the heathland plants after the application of fertilisers and wastewater is in contrast to the 'phosphorus toxicity' seen by Grundon (1972), Heddle and Specht (1975), Specht (1975), Groves and Keraitis (1976), and Ozanne and Specht (1981). It is also in contrast with the results of Dutch and Wolstenholme (1994), Stewart and Flinn (1984) and Lehane (1995), who found the application of sewage wastewater resulted in plant death, growth reduction or deformity. The one exception to this lack of death or disease — the discolouration of *Banksia marginata* leaves after the application of fire-fighting foam to unburnt vegetation — may not necessarily be due to the addition of nutrients but could have been the result of surfactants and other chemicals in the foam.

The absence of major detrimental effects was probably due to the small amounts of nutrients added as fertiliser or in the wastewater, and the relatively short period of time the nutrients were added. These experiments therefore indicate that not all accessions of nutrients result in adverse impacts on heathlands. This conclusion comes with the proviso that there may not have been time for some impacts to become obvious, the longest experimental monitoring period being only two years. Changes to plant biomass, canopy size or projective cover can take from 3 to 22 years to become significant (Specht 1963; Jones 1968; Heddle and Specht 1975; Hobbs and Atkins 1991). It is also possible that there were some responses to nutrient additions that were not measured, such as variations in the number of seeds germinating or changes in below-ground biomass.

Although there were no dramatic and immediate negative responses, there were some measurable increases and decreases in canopy size and live plant cover. These changes were found in all three experiments and after fertilisation in the heath–pasture boundary survey. Canopy increases from wastewater or fire-fighting foams can be attributed to either the addition of nutrients or the addition of water, or both (Grigg *et al.* 2000). Both experimental sites have sandy soils that drain freely, meaning that excess water becomes unavailable relatively quickly. Given that some of the canopy increases occurred more than one season after the applications of wastewater or foam plus water, the cover increases were probably not solely due to the addition of water. Conversely, the small amount of nutrients added may not have been enough, in themselves, to result in the increases of cover that were measured. The combination of both water and nutrients is postulated to have caused the increases in plant cover and growth found in these two experiments. The short time-scale of the experimental periods again creates a limitation to the interpretation of the data. That is, it is not possible to tell if the increased growth is part of a speeded up life cycle as found by Heddle and Specht (1975) or a purely short-term phenomenon.

Data from the heath–pasture boundaries survey shows that the level of nitrogen and phosphorus in the soil at the boundary can affect the cover of native plants, although it was not always easy to relate these changes back to the fertilising regime of the pasture. There was a general correlation between a higher phosphorus-to-nitrogen ratio at the edge of the heathland and more native cover at the edge. At some sites, such as those on private land, there was a distinct discontinuity between the relatively tall closed canopy of heathland shrubs and the low herbaceous pasture, forming a 'hedge' of plant growth as described by Grigg *et al.* (2000). However, there were no specific fertilising regimes that appeared to encourage such increased native cover. The application of fertilisers using a superphosphate spreader appeared to benefit native graminoids at the heath edge. This result could be attributed to the higher nitrogen levels also measured at the edge, disturbance from the super spreader reducing the competitive ability of shrubs and

increasing that of graminoids, or some other environmental factor (such as the soil water balance) not measured. Nitrogen seemed to play a bigger part than phosphorus in increasing native cover. High nitrogen levels were found in conjunction with the following: high native grass cover for flat areas and when the pasture was grazed all year; high total native cover when there were short periods between when the pasture was fertilised; high shrub cover when the pasture had been sown more than once or for crown land; and high graminoid cover when the stocking rate in the pasture was high. Other research into the effects of nutrient additions on heathland has shown much more dramatic increases in native shrub growth or cover (Specht 1963; Clark 1975; Heddle and Specht 1975; Specht *et al.* 1977).

Negative impacts seen in the present study include decreases in cover and canopy growth for certain species after wastewater was used to irrigate the heath and after fire-fighting foam was added. Cover values of *Aotus ericoides* (Golden Pea), *Baeckea ramosissima* (Baeckea or Rosy Heath Myrtle) and *Bossiaea cinerea* (Showy Bossiaea) declined more or increased less over time when low levels of fertilisers were added. Fire-fighting foam also caused leaf damage to *Banksia marginata* (Honeysuckle or Silver Banksia). From the heath–pasture boundaries survey, relatively low levels of canopy cover were measured for native grasses at the boundary with pasture that was on reserved land, on lighter soils and in the absence of sheep. In the first case phosphorus levels were high and in the other cases nitrogen levels were high. In most other situations in the survey and the experiments, native grasses increased or had higher cover when soil nutrient levels were higher. It is unclear why they acted differently in these situations, unless it was due to the soil pH, which was more acidic. There was a general association between lower native grass cover and lower pH in different circumstances seen during the survey. Given that the proportion of herbaceous species is generally low in heathlands, the canopy responses of native grasses are not indicative of a widespread effect on heathland condition. Also observed in the survey was lower native cover (particularly herb cover) for flatter slopes, which had higher phosphorus levels. There are many examples from other heathland research where growth has been reduced or there was no response at all (Specht 1963; Grundon 1972; Heddle and Specht 1975; Specht 1975; Groves and Keraitis 1976; Specht *et al.* 1977; Ozanne and Specht 1981). The individual plant responses to the addition of nutrients would have to be investigated at the species level to tease out the real relationships between phosphorus and changes in native cover.

Such negative impacts are similar to those seen by some researchers after applying large amounts of fertilisers to heathlands (Connor and Wilson 1967; Aerts and Berendse 1988) and after applying fire-fighting foams (Norecol 1989; Larson and Newton 1996). These canopy decreases show that different species can have quite different responses to the addition of nutrients. Over time, this could result in changes to the dominance

relationships or community composition in the heathland. It is apparent that most of the negative responses occurred due to the presence of fire-fighting foam rather than from wastewater or low levels of fertilisers. The detrimental effects of the foam could be caused by the nutrients, surfactants or other chemicals in the foam, the growth characteristics of the species and/or changes in soil nutrients resulting in an imbalance of trace elements.

The specific weather conditions during a nutrient accession event can have a bearing on subsequent impacts. Changes in canopy health (most often leaf damage but occasionally increased health) seen in the present study after the addition of fire-fighting foam were usually delayed by at least one season. The effects of both fire-fighting foams and fire retardants are known to be influenced by the post-treatment weather conditions (Bradstock *et al.* 1987; Norecol 1989; Larson and Newton 1996; Hamilton *et al.* 1998). The impacts of wastewater can also be influenced by the weather. Soluble elements in the wastewater, including salt, will have less effect if quickly leached out of the soil profile by ensuing rainfall. The length of time phosphorus is retained in heathland soils can vary, depending on the soil type, uptake by the vegetation and the amount of water entering the system (Ozanne *et al.* 1961; Heddle and Specht 1975; Witkowski *et al.* 1990a). In the present study, phosphorus seemed to be retained over the two years of measurement in the fire, fertilisers and weed seeds experiment, and over the 20 months of the wastewater experiment. Levels in both cases were significantly higher at the end of the experiment compared with the beginning. In contrast, the survey of heath–pasture boundaries showed that available phosphorus levels did not remain high in the soil for more than one year.

The implications of these results for the conservation and management of heathlands depends on how much the changes in growth and canopy cover affect the species composition of the heathland in the future. Faster growing species may become more dominant in the community and slower growing species less dominant or even absent. Alternatively, faster growing species may have a speeded up lifecycle that results in their death earlier than usual. Most of the changes in growth and flowering were minor enough, over the course of the study, for there not to be a consequential impact on the community composition. Although age of heath–pasture boundaries was shown not to be a contributing factor to a decline in heathland condition, a return in 10–20 years time to the three experimental sites would provide the answer as to whether any of the responses measured in the present study persist in the longer term.

Effects on species richness and community composition

The data presented in the present study show that there was no decrease in total or lifeform group species richness after the addition of fertilisers, wastewater or fire-fighting foam. Nor were there major changes to the heathland species composition. This is

in direct contrast to evidence provided by Specht (1975), Specht *et al.* (1977) and Specht and Specht (1989) showing that heathland vegetation is highly sensitive to nutrient changes. These results suggest that heathlands, at least in Tasmania, are relatively robust and can remain in good condition despite small inputs of nutrients. 'Good condition' in this case means maintaining the native species richness and remaining free of exotic invaders.

Although this is the overall picture, the heath–pasture boundary survey did show small declines in heathland condition — weed invasion in particular — related to relatively high levels of both soil nitrogen and phosphorus. Phosphorus was positively correlated with most of the exotic cover, richness and lifeform variables, particularly at the heath boundary. More acidic soils were also shown to be associated with good heathland condition. These data support a link between high nutrient levels and increased weed invasion as shown by Specht (1963), Connor and Wilson (1967), and Heddle and Specht (1975) for heathlands and Hobbs and Atkins (1988), Cale and Hobbs (1991) and Hester and Hobbs (1992) for other vegetation types in Australia. However, as stated above, Tasmanian heathlands appear to be more robust and resistant than might be expected given the level of disturbance.

The experiments and survey showed other cases where minor declines in heathland condition occurred in apparent response to nutrient additions. These impacts are considered minor for a few different reasons. The significantly increased frequency of exotic species in the fertiliser, fire and weed seed experiment was short-lived, lasting either one season or a few years only. The reduction in the number of native grass species in foamed plots in the fire-fighting foam experiment was also short term, and could have been a function of the small number of grasses found in that particular heathland community. In the survey, native richness increased as the number of years between fertilisation of the pasture increased, whereas phosphorus decreased. However, the data do not indicate if the fertilisation regime causes the changing phosphorus levels or if the changing phosphorus levels cause the changes in species richness. It is postulated that there is an association between them. In some cases, native richness was actually higher when phosphorus levels were also higher. More research would have to be done to investigate whether such changes are long term and thus how they would contribute to changes in heathland condition.

The null hypothesis, that realistic applications of nutrients to heathlands have no effect on their species richness, composition and growth, needs to be rejected on the grounds that some detrimental effects were measured. The addition of small amounts of nutrients obviously does nothing to improve the heathland condition. However, the present study shows that realistic amounts of nutrients (on their own) do not have a large negative effect, at least in the short term. This has positive implications for the conservation of heathlands because it means that small islands or fragments of heath subject to nutrient

accessions are still worth conserving as they appear unlikely to lose native species or be overwhelmed by the invasion of exotic species.

THE EFFECTS OF SMALL AMOUNTS OF NUTRIENTS COMBINED WITH FIRE

In two of the experiments in the present study, fire was included in the experimental design. In both the fertiliser and fire-fighting foam experiments, the fertiliser and foam were added after the burn had taken place in order to both maximise any possible impacts and to simulate what would usually occur through human activity. Fire was not included in the wastewater monitoring experiment because there was not the same level of control over the site or situation. Information about fire history was gathered during the heath–pasture boundaries survey, where it was available.

Effects on growth and flowering

From the evidence of other researchers, one would expect a fire to amplify any effects of nutrient additions by exacerbating the 'phosphorus toxicity' effect. That is, the imbalance between phosphorus and nitrogen that occurs when phosphorus fertilisers are added to heathland would be made worse when burning volatilises the available nitrogen (Specht 1963; Grundon 1972; Heddle and Specht 1975; Specht 1975; Groves and Keraitis 1976; Siddiqi *et al.* 1976b; Specht 1976; Ozanne and Specht 1981). Decreased canopy growth, health or flowering are all possible responses to phosphorus toxicity.

Comparison between the different experiments shows two responses to the combination of burning with the addition of nutrients in their different forms. In the first case, there were no interactive effects between fire and fertilisers on the cover of plants in the fertiliser, fire and weed seed experiment. Nor was there any major effect of fire and foam on the canopy health of five common heath shrubs in the fire-fighting foam experiment. The lack of a measurable response from nutrients and fire is not surprising given there was no great impact from the addition of nutrients alone. This indicates that the combination of small amounts of nutrients with a disturbance in the shape of fire is not enough to have a harmful effect on heathland condition. This is particularly surprising for the fire, fertilisers and weed seeds experiment because these two edge disturbances, fire and fertilisers, occurred in conjunction with the deliberate sowing of exotic grass seeds and grazing by introduced herbivores (rabbits). This site also had the added disturbance of a high fire frequency in the two decades prior to the present study.

In the second case, there were some relatively minor responses to fire-fighting foams and fire that were either negative (reduced canopy growth of *Banksia marginata* and reduced flowering of *Leptospermum scoparium*) or, in one case, positive (increased canopy growth of

Aotus ericoides). These responses could be due to changes to the soil nutrients or due to surfactants or other chemicals in the foam. They are considered relatively minor responses because they were generally short-term, in some cases the shrub canopies returning to 'normal' (not significantly different to the control) before the end of the experiment. The reduction in flowering in *L. scoparium* needs to be investigated further to ascertain whether it is a simple reduction in the numbers of flowers, a change in the way nutrients are allocated between growth and flowering or a delay or acceleration in the timing of flowering. An advancement of flowering by 1–2 weeks after application of phosphorus fertilisers was seen by Specht (1963). *L. scoparium* is a very common heathland species, not only at the experimental site in the Mt William National Park, but throughout Tasmania. A minor reduction in flowering over one season would therefore not threaten the viability of this species overall.

As with the effects of nutrient additions alone, the effects of nutrients and fire combined would depend on whether the changes in growth and canopy cover of different species caused a long-term change in the community composition and ecology. For the species whose canopy changes remained significantly different at the end of the experiment, the implications for conservation would depend on their continued presence or absence from the heathland community. Again, a return in 10–20 years time to the experimental sites would show whether any of the responses measured in the present study remain in the longer term.

Effects on species richness and community composition

The evidence from the present study on the effects of fire and small amounts of nutrients on species richness and heathland community composition provides limited support for the results seen in the literature. That is, when fire-fighting foam is applied to heathland that has been recently burnt there is a short-term reduction in native species richness (Larson and Newton 1996). However, no such reduction was evident when fertilisers were applied to recently burnt heathlands in the present study. There was even a positive impact of fire and foam, seen as a short-term increase in native grass species. The 'phosphorus toxicity' effect was thus not apparent in the present study.

The reduction in species richness induced by the foam and fire was confined to graminoids, herbs and orchids, in other words the understorey beneath the shrub layer. These decreases, combined with the increase in grass species, have the potential to lead to a significant change in the ecology of the heathland. What is encouraging in terms of conservation is that these changes in species richness were short term, occurring in different seasons and quickly returning to the same level as the unfoamed plots.

Species richness in heathlands is usually directly related to burning, decreasing as the time since fire decreases. Some researchers have found that not to be the case in heathlands they have studied (Purdie 1977a; Bradstock *et al.* 1997), and this is true also for heathlands in the present study. In the fire, fertilisers and weed seeds experiment, and in the fire-fighting foam experiment, there was no fire-induced increase in native species richness. This was regardless of the treatments being applied. Although the floristic composition was generally the same after fire, the relative dominance of species was different, reflecting their different regeneration strategies, a situation also documented by Purdie (1977a).

Although there were some responses to foam and fire, there was no increase in exotic species, and no interactive effects between fire and fertilisers in the fire, fertilisers and weed seeds experiment. It appears that these heathlands can maintain their condition even when there is a certain level of disturbance. I suggest that the addition of other edge disturbances, such as physical disturbance of the soil or a higher level of grazing or a much higher level of added nutrients, would eventually tip the balance and lead to invasion by exotic species, loss of native species and resultant change to the heathland structural and floristic community.

The null hypothesis, that realistic applications of nutrients to heathlands, in interaction with fire, have no effect on their species richness, composition and growth, needs to be rejected, again on the grounds that some detrimental effects were observed. However, most responses were minor and short term, becoming insignificant by the end of the study. Realistic amounts of nutrients combined with fire did not cause a major decline in heathland condition.

THE EFFECTS OF OTHER TYPES OF DISTURBANCE

Weed invasion

There are three different approaches to weed invasion used in the present study. Firstly, weed invasion is used as an indication of the condition of the heathland, a result of other types of disturbance. In this sense weed invasion has already been discussed in the previous two sections. Secondly, weed invasion is used as a treatment, i.e. applying weed seeds to quadrats in the fire, fertilisers and weed seeds experiment and sowing seeds in pasture next to heathland as described in the survey. This sense of weed invasion is discussed in the Sowing section below. Thirdly, weed invasion is a disturbance in its own right, and it is in this sense that it is discussed in this section.

All sites in the present study are potentially subject to weed invasion. The Peter Murrell Nature Reserve (site of the fire, fertilisers and weed seeds experiment), is surrounded by

suburbs, a golf course, an industrial area and a pony club. It is probably the most disturbed of all the sites in the present study, with the movement of people, dogs, horses and vehicles common along tracks through the heath. Rubbish dumping and trail bike riding occurred regularly before the site was declared as a Nature Reserve and the results of these disturbances are obvious in eroded tracks and piles of rubble. Despite these edge effects, the level of exotic species in the heath was low, with only two exotic species being found in the quadrats, other than the exotic grass seeds that were sown. Although the frequency of quadrats containing one of the exotic species increased over the course of the experiment, one year later the frequency had decreased again. The next most disturbed site is Stieglitz (site of the wastewater monitoring experiment), which is within one kilometre of shacks and gardens. It is also subject to some vehicle traffic directly through the heath, people walking and trail bike riding nearby. In the recent past there was the soil disturbance associated with creating the sewage ponds and the burial of pipes from the ponds to the nearby aerodrome. Despite these potential weed sources, no exotic species were found. Mt William National Park (site of the fire-fighting foam experiment), is probably the least disturbed, having the ocean on one side and large areas of national park on the other three sides. There is some walking and vehicle movement but only on formed tracks. Again, despite these potential weed sources, there were only three exotic species identified over the course of the experiment.

Weed invasion into heathland was more obvious in the heath–pasture boundaries survey. Certain types of management and forms of disturbance were associated with more exotic species. In contrast to the findings of Saunders *et al.* (1987) and Gilfedder and Kirkpatrick (1998), publicly owned land (national parks and crown land) had higher levels of exotics at the edge than privately owned heathland. However, the land tenure category of both the heathland and the pasture was a poor indicator of the disturbance levels or soil nutrients at the boundary. Regular slashing and clearing inside the heathland resulted in more weed invasion. There was a general association between disturbances in the heath (tracks, grazing, animal diggings, broken or fallen vegetation, wind pruning, rubbish dumping and soil disturbance) and higher levels of exotic species richness and cover. And, finally, more weeds were apparent in the absence of *Phytophthora cinnamomi*, discussed below.

One feature seen during the heath–pasture boundaries survey may actually retard the invasion of exotics. The hedge of increased growth of native species at the boundary found at some sites may act to reduce the amount of weed seeds reaching the heath centre. A relationship is suggested between cattle grazing in the pasture, low native cover at the heath edge and deeper weed penetration into the heath centre. Other results from the survey were equivocal in this regard, but do suggest this as a mechanism. Such shielding from weed invasion has been shown for long-standing forest edges (Ranney *et al.* 1981) and

at the edge of a temperate deciduous forest (Matlack 1993). Weed species that prefer a high light environment and some disturbance can have difficulty establishing in the thick growth found at the edge (Burke and Nol 1998). Exotic annuals have been shown to have difficulty establishing under the intact canopy of heathland (Hobbs and Atkins 1991).

The experiments show that low levels of nutrients added to heathlands, even when combined with fire and low levels of grazing, are not enough to result in a major increase in exotic species. However, the survey shows that increased nutrient levels are, on the whole, associated with higher weed invasion. Although nutrient increases did often take place in conjunction with grazing, trampling, soil disturbance and other types of disturbance, it was not possible to know all types and occurrences of disturbance to the heath prior to and during the survey. Thus, the data on weed invasion from the experiments and the survey are not in conflict with the idea that there is a threshold of disturbance for maintaining heathland condition. Other researchers have found a combination of disturbance important for weed invasion. For example, increased fire frequency plus nutrient-laden dust (Bridgewater and Backshall 1981), soil disturbance combined with high nutrient additions (Hobbs and Atkins 1988) and fire and soil disturbance (Hobbs and Atkins 1991).

Grazing

Although grazing is generally not considered a major threat to Australian heathlands due to the density of the shrub cover and lack of palatable species (Specht 1981a; Hobbs 1991), the heath–pasture boundaries survey showed that grazing can impact negatively on heathland condition. In fact, there were more responses by exotic variables to the grazing regime than there were to the fertilising or sowing regimes. Domestic stock can encourage weed invasion in a variety of ways, such as adding nutrients in their faeces (Hobbs 1987; Hobbs 1991; Scougall *et al.* 1993; Pettit *et al.* 1995; Fox *et al.* 1997; Yates *et al.* 2000) or physically trampling native plants and reducing their biomass (Leigh and Holgate 1979). Both mechanisms appeared to be in operation in the survey. For nutrient enrichment, relationships were seen between high nitrogen levels and high exotic richness at the heath edge when cattle grazed in the pasture; and between high nitrogen levels, high exotic richness and grazing for less than half the year. For physical trampling and destruction of the native canopy, relationships were seen between the presence of cattle in the pasture and a decline in native cover at the heath edge; and between stock grazing regularly in the heathland and a decline in native cover in the heath centre. An intact heathland canopy has been shown to hinder weed invasion (Hobbs and Atkins 1988; Hester and Hobbs 1992), thus any reduction in canopy cover can encourage weed invasion (Hobbs and Atkins 1991).

Continued grazing in the heathland could conceivably worsen the heathland condition. Stock selectively graze palatable native species that may not be able to recover from the defoliation, whereas some exotic species (such as those with unpalatable prickles and thorns) are avoided (Amor and Piggin 1977). However, in the survey the opposite was seen, that is, domestic stock appeared to selectively graze exotic species when they were present, and it was not until some years after they left that exotic species were able to increase their cover and numbers. This was seen for weed penetration that was deepest for grazing in the pasture for 1–5 years rather than for current grazing, and for exotic herb cover that was highest when grazing had occurred more than 5 years ago compared with more recent grazing. Although not seen specifically in the survey, the conjunction of the physical disturbance of grazing, the introduction of exotic species and burning could also negatively impact the heath if the exotics were fire tolerant. Given these problems, fencing to keep out domestic stock would obviously make a difference in conserving heathland in good condition. This suggestion is supported by the connection between more barriers (fences, ditches etc) at the heath boundary and fewer weeds in the heathland that was observed in the survey. The disadvantage of fencing is, of course, its high cost (Hobbs *et al.* 1993).

Native and feral herbivores are known to preferentially graze on the regenerated green shoots available in recently burnt heathland (Specht 1981a). This was obvious from an increase in scat numbers in the burnt areas of both the fire, fertilisers and weed seeds experiment and the fire-fighting foam experiment. However, at these two sites there was no observable interaction between the fertiliser or foam treatment and scat numbers. It is possible that there was an interaction between herbivory and the failure of exotic grass species to persist in the fire, fertilisers and weed seeds experiment, although this was not directly measured. In contrast, native herbivores were found to encourage weed invasion in the heathlands studied in the survey. The presence of exotics was found to increase with the presence of medium to high numbers of wombats and wallabies, Cape Barren Geese grazing in the pasture, and animal diggings in the heathland. The mechanism for this increase in weeds appeared to be the hard, bare trails that the larger native herbivores (in particular wombats, wallabies and pademelons) made into the heathland. Weed numbers were found to be higher on the trail compared with the heath to either side, and to decrease with distance along the trail.

Sowing

The establishment of deliberately sown exotic species was very low at the two experimental sites where a sowing treatment was applied. In the fire, fertilisers and weed seeds experiment, the exotic grass seeds germinated and were slightly more frequent when fertilised, but failed to persist despite adequate rainfall. This is particularly interesting

considering that one of the species broadcast, *Dactylis glomerata* (Cocksfoot), was found to be associated with other weeds at the heath boundary in the survey. In the waste water experiment, no seedlings were found from the exotic grass and non-local native tree seeds that were sown, despite being provided with water and nutrients from the sewage wastewater. It is quite likely that, at both sites, some level of soil disturbance was required to allow the exotic species to establish, as noted by other researchers (Hobbs and Atkins 1988; Hobbs 1989).

Results from the survey are similar, in that there was no clear evidence that the sowing regime in pasture adjacent to heathland had a great detrimental effect on the condition of the heathland. There did appear to be an association between the presence of two pasture grasses, *Dactylis glomerata* (Cocksfoot) and *Festuca* species (Fescue), and increased exotic herb cover, grass cover and species richness in the heathland. However, given the difficulty in identifying the exotic grasses that were found in the heathland and the variation in exotic cover between the different regions covered by the survey, it is not possible to state that these pasture grasses are aggressive invaders of heathlands. It is quite likely that they are actually indicators of a set of environmental or management conditions that are responsible for weed invasion. Autumn sowing also appeared to result in more weed invasion into the heath, although there could have been other contributing factors such as the fertilising regime and hence soil phosphorus levels. Exotic grasses can invade heathlands (Amor and Stevens 1975; Wace 1977; Rose and Webb 1994; Mendoza *et al.* 1995), but it appears from this survey that they require some level of disturbance (nutrient additions or soil disturbance) to enable them to seriously challenge heathland integrity.

Phytophthora cinnamomi

Phytophthora cinnamomi was not obviously active at two of the study sites. However, at the fire, fertilisers and weed seeds experimental site, the pathogen was present and widespread. Many susceptible species, such as *Sprengelia incarnata* and other epacrids, were either not present or present and experiencing dieback (Kirkpatrick and McQuillan 1996). Plants vulnerable to the fungus can quickly become absent from the community, and there is often a rapid decline in shrub species and increase in sedges and rushes (Weste and Law 1973; Weste 1974). Species richness at the fire, fertilisers and weed seeds site was low compared with the other two sites and many of the heathlands visited during the survey.

A severe *Phytophthora* invasion has the potential to create the right conditions for weed invasion by causing a reduction in species richness and cover of susceptible native species. However, the opposite was found in the survey; that is, the presence of the fungus was associated with less weed invasion. The three possible reasons for this finding are, firstly,

that the conditions favouring *Phytophthora* invasion do not favour weed invasion, secondly, that *Phytophthora* had already passed through the heathland before the survey took place, resulting in a decline in heathland condition and hence weed invasion and, thirdly, *Phytophthora* could have been present but without obvious symptoms. The first explanation is considered mostly likely, for the following reasons. The soil pH measured in the field was more acidic when *Phytophthora* was present, supporting the known conditions for *Phytophthora* invasion, i.e. acidic shallow soils with poor drainage (Weste 1974). Many pasture species and other exotics would prefer well drained, more neutral soil and hence would be unlikely to become established in the sites where *Phytophthora* was present. Another finding from the heath–pasture boundaries survey that supports this explanation is that the acidity of the soil plays a big part in the presence of weeds in heathlands generally.

Physical disturbance

Observations from all three experiments and the survey support the evidence that heathland vegetation is relatively fragile and easily damaged by physical disturbance (Buchanan 1979; Soulé *et al.* 1992; Rose and Webb 1994). In the wastewater experiment and in the fire-fighting foam experiment, crushing of plants and compaction of soil due to vehicles driving through the heathland remained obvious (although not specifically measured) for the entire study period. The quadrats in the fire, fertiliser and weed seeds experiment were originally hard to find as they were marked only with 20 cm long metal deck spikes in each corner. However, by the end of the experimental period they were quite visible as trampled squares around the outside of each edge, despite all care being taken. The survey showed a decline in native species richness in relation to soil disturbance at the heath–pasture boundary. The survey also showed that increasing amounts of physical disturbance resulted in a decline in the heath condition, with exotic species richness and exotic cover increasing, and native cover decreasing. The increasing amounts of physical disturbance consisted of an increase in the number of different types of disturbance, supporting the idea of a threshold of disturbance beyond which a loss of heath condition occurs.

The null hypothesis, that the nature of the disturbance regime on both sides of the heath–pasture boundary has no effects on heathland species richness, composition and growth, can clearly be rejected. Weed invasion, sowing, grazing and physical disturbance all impact negatively on heathland condition. The pathogen *Phytophthora cinnamomi* appears to be inversely correlated with weed invasion, probably an artefact of its virulence on the more poorly drained and infertile soils. The combination of several

different types of edge effects, such as grazing with nutrient increases or sowing with physical disturbance, is more likely to result in a decline in heathland integrity than each edge effect alone.

CONSERVATION OF HEATHLANDS

The impacts on heathland of small amounts of nutrients alone and with fire, and the impacts of other types of disturbance, were shown to be minor and short term. The evidence from the present study is that heathlands are relatively robust and that small fragments of heath will not be overwhelmed by edge effects. This evidence supports the view that small remnants of native vegetation are not necessarily all 'edge' (Burke and Nol 1998) and are worth conserving (Ogle 1987).

Many remnants, particularly on private land, are managed by 'benign neglect', but a more educated approach is needed to manage such remnants to preserve both species and communities into the future. When conserving small fragments of heathland, an approach suggested by this research is to observe the overall load of disturbance from all possible sources. Some types of disturbance, such as large increases in nutrient levels or heavy grazing presence by domestic stock and/or native mammals, may be enough in themselves to result in a loss to heathland condition. Other types of disturbance, for example pasture sowing or the addition of small amounts of nutrients, may be able to be tolerated by the heathland. What appears to be important is a threshold of disturbance. If a site was particularly vulnerable or valuable in terms of rare or threatened species, all disturbances could be considered detrimental to be on the safe side. However, small amounts of disturbance or a combination of different types of disturbance could be tolerated in other sites.

The present study does suggest some management actions that could be taken to reduce the impact of disturbance edge effects. To counteract the accession of high levels of nutrients into the heathland, a buffer could be left between the pasture and the heath boundary. This could be achieved by leaving a strip around the heathland where phosphorus fertilisers and lime were not applied. Weed invasion in the survey dropped markedly after 20 m, so this could be an adequate width for a buffer strip. Buffer zones around heathland have been suggested by many researchers as a way to limit negative impacts of surrounding agricultural land (Specht and Cleland 1961; Angold 1997; Bowler 2000). To counteract the effects of increased nutrients leaching from roads or tracks Specht (1981a) recommends putting down a layer of lateritic soils with high sorption capacity for phosphorus. Such a layer could be used around a particularly valuable heathland remnant, if the cost of such a buffer was not prohibitive.

A buffer zone would also be useful to counteract the invasion of weeds into a heathland. As well as not fertilising in the buffer zone, not sowing exotic nitrogen-fixing legumes or pasture grasses in this zone would help reduce weed invasion. Other management practises that would reduce weed invasion would be to keep cattle out of the heathland and, if possible, fence the heath–pasture boundary. If a heathland remnant were particularly valuable and cost was not a factor, fencing out wallabies and wombats would also reduce weed invasion. The restriction of physical disturbances, such as rubbish dumping, slashing, trampling and cutting of shrubs, would obviously aid considerably in preserving heathland condition. Hobbs and Atkins (1991) suggest maintaining intact vegetation canopies as the most effective way of preventing the invasion of exotic species into sandplain heath in semi-arid Australia. Fire does not seem to encourage weed invasion in the heathlands studied and this is also the case in other heathlands in Australia (Hobbs and Atkins 1991). However, maintaining an appropriate fire frequency would be part of reducing the threshold of possible disturbances.

Future research into heathland conservation is needed to clarify at what level of nutrients the impacts become unacceptable. The present study provides a lower limit, and other research has supplied upper limits, but there is a large range in between. Further research into the effects of wastewater and fire-fighting foam on heathland would be useful to separate out the effects of water, of nutrients and of other constituents in the foam or wastewater. The survey of heath–pasture boundaries throws up almost as many questions as answers for heathland conservation. More detailed assessment of different species' response to different disturbances would be productive, as would a larger experiment combining a range of different types of disturbances in a factorial design, including soil disturbance.

7. References

Abbott, I. (1985). Rate of growth of *Banksia grandis* Willd. (Proteaceae) in Western Australian forest. *Australian Journal of Botany* **33**: 381–391.

Adam, P., Stricker, P. and Anderson, D.J. (1989). Species-richness and soil phosphorus in plant communities in coastal New South Wales. *Australian Journal of Ecology* **14**: 189–198.

Adam, P., Stricker, P., Wiecek, B.M. and Anderson, D.J. (1990). The vegetation of seacliffs and headlands in New South Wales, Australia. *Australian Journal of Ecology* **15**: 515–547.

Adams, M.A., Iser, J., Keleher, A.D. and Cheal, D.C. (1994). Nitrogen and phosphorus availability and the role of fire in heathlands at Wilsons Promontory. *Australian Journal of Botany* **42**: 269–281.

Adams, R. (2000). Ecological Effects of Class A Foams and their Potential Ecological Impacts on Australian Ecosystems. February 2000. Country Fire Authority Victoria, Melbourne.

Adams, R. and Simmons, D. (1999). Ecological effects of fire fighting foams and retardants: a summary. *Australian Forestry* **62**(4): 307–314.

Aerts, R. (1990). Nutrient use efficiency in evergreen and deciduous species from heathlands. *Oecologia* **84**(3): 391–397.

Aerts, R. and Berendse, F. (1988). The effect of increase of nutrient availability on vegetation dynamics in wet heathlands. *Vegetatio* **76**: 63–69.

Aerts, R., Berendse, F., de Caluwe, H. and Schmitz, M. (1990). Competition in heathland along an experimental gradient of nutrient availability. *Oikos* **57**(3): 310–318.

Amor, R.L. and Piggin, C.M. (1977). Factors influencing the establishment and success of exotic plants in Australia. *Proceedings of the Ecological Society of Australia* **10**: 15–26.

Amor, R.L. and Stevens, P.L. (1975). Spread of weeds from a roadside into sclerophyll forests at Dartmouth, Australia. *Weed Research* **16**: 111–118.

Angold, P.G. (1997). The impact of a road on adjacent heathland vegetation — effects on plant species composition. *Journal of Applied Ecology* **34**(2): 409–417.

Angus Fire Armour (1997). Chemwatch Material Safety Data Sheet, Angus ForExpan-S Class A Foam Concentrate. Angus Fire Armour.

Auld, T.D. (1990). The survival of juvenile plants of the resprouting shrub *Angophora hispida* (Myrtaceae) after a simulated low-intensity fire. *Australian Journal of Botany* **38**: 255–260.

Barker, P.C.J. and Wardlaw, T.J. (1995). Susceptibility of selected Tasmanian rare plants to *Phytophthora cinnamomi*. *Australian Journal of Botany* **43**: 379–386.

Beadle, N.C.W. (1954). Soil phosphate and the delimitation of plant communities in eastern Australia. *Ecology* **35**: 370–375.

Benson, D.H. and Howell, J. (1990). Sydney's vegetation 1788–1988: utilization, degradation and rehabilitation. *Proceedings of the Ecological Society of Australia* **16**: 115–127.

Berendse, F. (1990). Organic matter accumulation and nitrogen mineralization during secondary succession in heathland ecosystems. *Journal of Ecology* **78**(2): 413–427.

Biodiversity Unit (1995). *Native Vegetation Clearance, Habitat Loss and Biodiversity Decline.* Biodiversity Series Paper No. 6. Department of the Environment, Sport and Territories, Canberra.

Boardman, R. (1991). Impacts on planted trees and soils of re-using effluents and waste waters in Australian environments — a review. Ed. Ryan, P.J., *Productivity in Perspective, Proceedings of the Third Australian Forest Soils and Nutrition Conference.* Forestry Commission, Sydney.

Bond, W.J. (1983). On alpha diversity and the richness of the Cape flora: a study in Southern Cape fynbos. Eds Kruger, F.J., Mitchell, D.T. and Jarvis, J.U.M., *Mediterranean-Type Ecosystems: The Role of Nutrients,* pp 337–356. Springer-Verlag, Heidelberg.

Bond, W.J. (1984). Fire survival of Cape Proteaceae — influence of fire season and seed predators. *Vegetatio* **56**: 65–74.

Bond, W.J., Midgley, J. and Vlok, J. (1988). When is an island not an island? Insular effects and their causes in fynbos shrublands. *Oecologia* **77**: 515–521.

Bowler, P.A. (2000). Ecological restoration of coastal sage scrub and its potential role in habitat conservation plans. *Environmental Management* **26**(suppl 1): S85–S96.

Bradstock, R.A. and Myerscough, P.J. (1981). Fire effects on seed release and the emergence and establishment of seedlings in *Banksia ericifolia* L.f. *Australian Journal of Botany* **29**: 521–531.

Bradstock, R.A. and Myerscough, P.J. (1988). The survival and population response to frequent fires of two woody resprouters *Banksia serrata* and *Isopogon anemonifolius*. *Australian Journal of Botany* **36**: 415–431.

Bradstock, R.A., Sanders, J. and Tegart, A. (1987). Short-term effects on the foliage of a eucalypt forest after an aerial application of a chemical fire retardant. *Australian Forestry* **50**(2): 71–80.

Bradstock, R.A., Tozer, M.G. and Keith, D.A. (1997). Effects of high frequency fire on floristic composition and abundance in a fire-prone heathland near Sydney. *Australian Journal of Botany* **45**: 641–655.

Bridgewater, P.B. and Backshall, D.J. (1981). Dynamics of some Western Australian ligneous formations with special reference to the invasion of exotic species. *Vegetatio* **46**: 141–148.

Bridgewater, P.B. and Zammit, C.A. (1979). Phytosociology of S.W. Australian limestone heaths. *Phytocoenologia* **6**: 327–343.

Brown, G. and Mitchell, D.T. (1986). Influence of fire on soil phosphorus status in sand-plain lowland fynbos, south-western Cape. *South African Journal of Botany* **52**: 67–72.

Buchanan, A.M., Ed. (1999). A Census of the Vascular Plants of Tasmania, 3rd edition. Tasmanian Herbarium Occasional Publication No. 6. Tasmanian Museum and Art Gallery, Hobart.

Buchanan, R.A. (1979). Edge disturbance in natural areas. *Australian Parks and Recreation* August: 39–43.

Buhl, K.J. and Hamilton, S.J. (1998). Acute toxicity of fire-retardant and foam-suppressant chemicals to early life stages of Chinook Salmon (*Oncorhynchus tshawytscha*). *Environmental Toxicology and Chemistry* **17**(8): 1589–1599.

Buhl, K.J. and Hamilton, S.J. (2000). Acute toxicity of fire-control chemicals, nitrogenous chemicals and surfactants to Rainbow Trout. *Transactions of the American Fisheries Society* **129**: 408–418.

Bureau of Meteorology (1999). Climate information. http://www.bom.gov.au. Australia.

Burgman, M. (1988). Spatial analysis of vegetation patterns in southern Western Australia: implications for reserve design. *Australian Journal of Ecology* **13**: 415–429.

Burke, D.M. and Nol, E. (1998). Edge and fragment size effects on the vegetation of deciduous forests on Ontario, Canada. *Natural Areas Journal* **18**(1): 45–53.

Cadenasso, M. L. and Pickett, S. T. A. (2000). Linking forest edge structure to edge function: mediation of herbivore damage. *Journal of Ecology* **88**: 31–44.

Cale, P. and Hobbs, R. (1991). Condition of roadside vegetation in relation to nutrient status. Eds Saunders, D.A. and Hobbs, R.J., *Nature Conservation 2: The Role of Corridors*, pp 353–362. Surrey Beatty and Sons, Chipping Norton, NSW.

Chapman, S.B. (1967). Nutrient budgets for a dry heath ecosystem in the south of England. *Journal of Ecology* **55**: 677–689.

Chapman, S.B., Rose, R.J. and Basanta, M. (1989). Phosphorus adsorption by soils from heathlands in southern England in relation to successional change. *Journal of Applied Ecology* **26**(2): 673–680.

Cheal, D.C. (1996). Fire succession in heathlands and implications for vegetation management. *Fire and Biodiversity: The Effects and Effectiveness of Fire Management*, Footscray, Melbourne, 8–9 October 1994, Biodiversity Series Paper No. 8. Biodiversity Unit, Department of Environment, Sport and Territories, Canberra.

Chen, J., Franklin, J.F. and Spies, T.A. (1992). Vegetation responses to edge environments in old-growth Douglas-fir forests. *Ecological Applications* **2**: 387–396.

Chen, J., Franklin, J.F. and Spies, T.A. (1995). Growing-season microclimatic gradients from clearcut edges into old-growth Douglas-fir forests. *Ecological Applications* **5**: 74–86.

Christensen, P.E. and Kimber, P.C. (1975). Effect of prescribed burning on the flora and fauna of south-west Australian forests. *Proceedings of the Ecological Society of Australia* 9: 85–106.

Christmas Hills Fire Brigade (2000). The ecological effects of foam. http://home.vicnet.net.au/~chfb/foam.html. Last revised 25 October 2000, Australia.

Clark, S.S. (1975). The effect of sand mining on coastal heath vegetation in New South Wales. *Proceedings of the Ecological Society of Australia* **9**: 1–16.

Clarke, P.J., Myerscough, P.J. and Skelton, N.J. (1996). Plant coexistence in coastal heaths: between- and within-habitat effects of competition, disturbance and predation in the post-fire environment. *Australian Journal of Ecology* **21**: 55–63.

Clemens, J. and Franklin, M.H. (1980). A description of coastal heath at North Head, Sydney Harbour National Park: impact of recreation and other disturbance since 1951. *Australian Journal of Botany* **28**: 463–478.

Clements, A. (1983). Suburban development and resultant changes in the vegetation of the bushland of the northern Sydney region. *Australian Journal of Ecology* 8: 307–319.

Cody, M.L. (1986). Diversity, rarity and conservation in Mediterranean climate regions. Ed. Soulé, M.E., *Conservation Biology, the Science of Scarcity and Diversity*, pp 122–152. Sinauer, Sunderland, Massachusetts.

Connor, D.J. and Wilson, G.L. (1967). Response of a coastal Queensland heath community to fertilizer application. *Australian Journal of Botany* **16**: 117–123.

Cook, F.J., Kelliher, F.M. and McMahon, S.D. (1994). Changes in infiltration and drainage during wastewater irrigation of a highly permeable soil. *Journal of Environmental Quality* 23(3): 476–482.

Coquillard, P. (1995). Simulation of the cyclical process of heathlands: induction of mosaic structures, evolution to irreversible states. *Ecological Modelling* **80**: 97–111.

Correll, D.L. (1991). Human impact on the functioning of landscape boundaries. Eds Holland, M.M., Risser, P.G. and Naiman, R.J., *Ecotones: The Role of Landscape Boundaries in the Management and Restoration of Changing Environments*, pp 90–109. Chapman and Hall, New York.

Costin, A.B., Gray, M., Totterdell, C.J. and Wimbush, D.J. (1982). *Kosciusko Alpine Flora*. CSIRO/Collins, Sydney.

Cowling, R.M., Lamont, B.B. and Enright, N.J. (1990). Fire and management of south-western Australian banksias. *Proceedings of the Ecological Society of Australia* **16**: 177–183.

Cromer, R.N., Tompkins, D. and Barr, N.J. (1982). Irrigation of *Pinus radiata* with waste water: tree growth in response to treatment. *Australian Forest Research* **13**: 57–65.

Curtis, N.P. (1998). A post-fire ecological study of *Xanthorrhoea australis* following prescribed burning in the Warby Range State Park, north-eastern Victoria, Australia. *Australian Journal of Botany* **46**: 253–272.

Davis, G.W. (1992). Effects of tillage on heathland soil: disturbance of a natural mountain fynbos system in the southwestern Cape, South Africa. *Soil and Tillage Research* **24**: 29–45.

DeBano, L.F., Eberlein, G.E. and Dunn, P.H. (1979). Effects of burning on chaparral soils: I. soil nitrogen. *Soil Science Society of America Journal* **43**: 504–509.

Debinski, D. M. and Holt, R. D. (2000). A survey and overview of habitat fragmentation experiments. *Conservation Biology* **14**: 342–355.

Department of Environment and Land Management (1994). Guidelines for Re-use of Wastewater in Tasmania. June 1994. Division of Environmental Management, Hobart.

Department of Lands, Parks and Wildlife (1987). Fire Management Plan Mt William National Park. August 1987. Department of Lands, Parks and Wildlife, Hobart.

di Castri, F. (1981). Mediterranean-type shrublands of the world. Eds di Castri, F., Goodall, D.W. and Specht, R.L., *Mediterranean-Type Shrublands*, pp 1–52. Elsevier, Amsterdam.

di Castri, F. (1991). An ecological overview of the five regions of the world with a mediterranean climate. Eds Groves, R.H. and di Castri, F., *Biogeography of Mediterranean Invasions*, pp 3–15. Cambridge University Press, Cambridge.

Diamond, J.M. (1975). The island dilemma: lessons of modern biogeographic studies for the design of nature reserves. *Biological Conservation* 7: 129–146.

Dickinson, K.J.M. and Kirkpatrick, J.B. (1986). The impact of grazing pressure in clearfelled, burned and undisturbed eucalypt forest. *Vegetatio* **66**: 133–136.

Dodge, M. (1970). Nitrate poisoning, fire retardants and fertilisers — any connection? *Journal of Rangeland Management* **23**: 244–247.

Duncan, D. and Duncan, F. (1995). Vegetation resources and management in the Coffee Creek Reserve and the Huntingfield Estate. Report to the Huntingfield/Coffee Creek Landcare Group, March 1995.

Duncan, M.J. and Keane, P.J. (1996). Vegetation changes associated with *Phytophthora cinnamomi* and its decline under *Xanthorrhoea australis* in Kinglake National Park, Victoria. *Australian Journal of Botany* **44**(3): 355–369.

Dunn, P.H., DeBano, L.F. and Eberlein, G.E. (1979). Effects of burning on chaparral soils: II. soil microbes and nitrogen mineralization. *Soil Science Society of America Journal* **43**: 509–514.

Dutch, J. and Wolstenholme, R. (1994). The effects of sewage sludge application to a heathland site prior to planting with Sitka spruce. *Forest Ecology and Management* **66**(1–3): 151–163.

Dwyer, P.D., Kikkawa, J. and Ingram, G.J. (1981). Habitat relations of vertebrates in subtropical heathlands of coastal southeastern Queensland. Ed. Specht, R.L., *Heathlands and Related Shrublands: Descriptive Studies*. Ecosystems of the World, Vol. 9A, pp 281–300. Elsevier, Amsterdam.

Enright, N.J. (1989). Heathland vegetation of the Spirits Bay area, far northern New Zealand. *New Zealand Journal of Ecology* **12**: 63–75.

Enright, N.J. and Lamont, B.B. (1989). Fire temperatures and follicle-opening requirements in 10 *Banksia* species. *Australian Journal of Ecology* **14**: 107–113.

Falkiner, R.A. and Polglase, P.J. (1997). Transport of phosphorus through soil in an effluent-irrigated tree plantation. *Australian Journal of Soil Research* 35: 385–397.

Feigin, A., Ravina, I. and Shalhevet, J. (1991). *Irrigation with Treated Sewage Effluent: Management for Environmental Protection*. Springer Verlag, Heidelberg.

Finger, S.E. (1995). Environmental implications of firefighting chemicals. *Foam Applications for Wildland and Urban Fire Management* 7(1): 1–6.

Finger, S.E., Poulton, B., Hamilton, S., Buhl, K., Vyas, N., Hill, E. and Larson, D. (1997). Toxicity of fire retardant and foam suppressant chemicals to plant and animal communities. Interior Fire Coordination Committee, Boise, Idaho, Final Report, 15/12/97.

Fox, B.J., Taylor, J.E., Fox, M.D. and Williams, C. (1997). Vegetation changes across edges of rainforest remnants. *Biological Conservation* **82**: 1–13.

Fox, M.D. (1995). Australian mediterranean vegetation: intra- and inter-continental comparisons. Eds Kalin Arroyo, M.T., Zedler, P.H. and Fox, M.D., *Ecology and Biogeography of Mediterranean Ecosystems in Chile, California and Australia.* Ecological Studies Vol 108: Analysis and Synthesis, , pp 137–158. Springer-Verlag, New York.

Fox, M.D. and Fox, B.J. (1986). The susceptibility of natural communities to invasion. Eds Groves, R.H. and Burdon, J.J., *Ecology of Biological Invasions: An Australian Perspective,* pp 57–66. Australian Academy of Science, Canberra.

Fraver, S. (1994). Vegetation responses along edge-to-interior gradients in the mixed hardwood forests of the Roanoke River basin, North Carolina. *Conservation Biology* **8**(3): 822–832.

Gehlhausen, S. M., Schwartz, M. W. and Augspurger, C. K. (2000). Vegetation and microclimate edge effects in two mixed-mesophytic forest fragments. *Plant Ecology* **147**: 21–35.

Gaikowski, M.P., Hamilton, S.J., Buhl, K.J., McDonald, S.F. and Summers, C.H. (1996). Acute toxicity of firefighting chemical formulations to four life stages of fathead minnow. *Ecotoxicology and Environmental Safety* **34**: 252–263.

Gellini, R.F., Pantani, P., Bussotli, F., Barbolani, E. and Rinallo, C. (1983). Survey of deterioration of coastal vegetation of in the Park of San Rossore in Central Italy. *European Journal of Forest Pathology* **13**: 2961304.

Gilfedder, L. and Kirkpatrick, J.B. (1993). Factors influencing the condition of natural vegetation remnants in subhumid Tasmania. Report to the Australian Nature Conservation Agency, Hobart, May 1993.

Gilfedder, L. and Kirkpatrick, J.B. (1998). Factors influencing the integrity of remnant bushland in subhumid Tasmania. *Biological Conservation* **84**: 89–96.

Gill, A.M. (1977). Management of fire-prone vegetation for plant species conservation in Australia. *Search* **8**: 20–25.

Gill, A.M. (1981). Adaptive responses of Australian vascular plant species to fires. Eds Gill, A.M., Groves, R.H. and Noble, I.R., *Fire and the Australian Biota,* pp 243–271. Australian Academy of Science, Canberra.

Gill, A.M. and Groves, R.H. (1981). Fire regimes in heathlands and their plant-ecological effects. Ed. Specht, R.L., *Heathlands and Related Shrublands: Analytical Studies.* Ecosystems of the World, Vol. 9B, pp 61–84. Elsevier, Amsterdam.

Gill, A.M. and Ingwersen, F. (1976). Growth of *Xanthorrhoea australis* R.Br. in relation to fire. *Journal of Applied Ecology* **13**: 195–203.

Gimingham, C.H. (1972). *Ecology of Heathlands*. Chapman and Hall, London.

Gosz, J.R. (1991). Fundamental ecological characteristics of landscape boundaries. Eds Holland, M.M., Risser, P.G. and Naiman, R.J., *Ecotones: The Role of Landscape Boundaries in the Management and Restoration of Changing Environments*, pp 8–30. Chapman and Hall, New York.

Griffin, E.A., Hopkins, A.J.M. and Hnatiuk, R.J. (1983). Regional variation in Mediterranean shrublands at Eneabba, south-western Australia. *Vegetatio* **52**: 103–127.

Grigg, A.M., Pate, J.S. and Unkovich, M.J. (2000). Responses of native woody taxa in *Banksia* woodland to incursion of groundwater and nutrients from bordering agricultural land. *Australian Journal of Botany* **48**: 777–792.

Groves, R.H. (1965). Growth of heath vegetation II. The seasonal growth of a heath on ground-water podzol at Wilson's Promontory, Victoria. *Australian Journal of Botany* **13**: 281–289.

Groves, R.H. (1981a). Heathland soils and their fertility status. Ed. Specht, R.L., *Heathlands and Related Shrublands: Analytical Studies*. Ecosystems of the World, Vol. 9B, pp 143–150. Elsevier, Amsterdam.

Groves, R.H. (1981b). Nutrient cycling in heathlands. Ed. Specht, R.L., *Heathlands and Related Shrublands: Analytical Studies*. Ecosystems of the World, Vol. 9B, pp 151–164. Elsevier, Amsterdam.

Groves, R.H. (1983). Nutrient cycling in Australian heath and South African fynbos. Eds Kruger, F.J., Mitchell, D.T. and Jarvis, J.U.M., *Mediterranean-Type Ecosystems: The Role of Nutrients*, pp 179–191. Springer-Verlag, Heidelberg.

Groves, R.H. and Keraitis, K. (1976). Survival and growth of seedlings of three sclerophyll species at high levels of phosphorus and nitrogen. *Australian Journal of Botany* **24**: 681–690.

Groves, R.H. and Specht, R.L. (1965). Growth of heath vegetation I. Annual growth curves of two heath ecosystems in Australia. *Australian Journal of Botany* **13**: 261–280.

Grundon, N.J. (1972). Mineral nutrition of some Queensland heath plants. *Journal of Ecology* **6**: 171–181.

Hahs, A., Enright, N.J. and Thomas, I. (1999). Plant communities, species richness and their environmental correlates in the sandy heaths of Little Desert National Park, Victoria. *Australian Journal of Ecology* **24**(3): 249–247.

Hamilton, S.J., Larson, D., Finger, S., Poulton, B., Vyas, N. and Hill, E. (1998). Ecological effects of fire retardant chemicals and fire suppressant foams. Jameston, ND: Northern Prairie Wildlife Research Centre Home Page, http://www.npwrc.usgs.gov/resource/OTHRDATA/Fireweb/FIREWEB.HTM. USA, version 02MAR98.

Hamilton, S.J., McDonald, S.F., Gaikowski, M.P. and Buhl, K.J. (1996). Toxicity of fire retardant chemicals to aquatic organisms: Progress report. Ed. Ramsey, G.S., *International Wildland Fire Foam Symposium and Workshop*, Thunder Bay, Ontario, 3–5 May 1994. Information Report PI-X-123, Natural Resources Canada, Canadian Forest Service.

Hammill, K.A., Bradstock, R.A. and Allaway, W.G. (1998). Post-fire seed dispersal and species re-establishment in proteaceous heath. *Australian Journal of Botany* **46**(3–4): 407–419.

Hanna, D., Shaw, S., Boardman, R., Schrale, G., Gram, S., Dillon, P. and Lauck, V. (1994). Nutrient, solute and water distribution in an Australian native hardwoods plantation irrigated with reclaimed water from Bolivar sewage treatment works. *Water Down Under '94: Proceedings of International Association of Hydrogeologists and Institute of Australian Engineers*, Adelaide, South Australia, 21–25 November 1994. International Association of Hydrogeologists and Institute of Australian Engineers.

Hansson, L. (2000). Edge structure and edge effects on plants and birds in ancient oak–hazel woodlands. *Landscape and Urban Planning* **46**: 203–207.

Hartskeerl, K. (1999). The Ecological Effects of Class A Foam on the Growth of Native Australian Terrestrial Plants. Honours Thesis, 61 pages. Melbourne, Deakin University.

Haseler, M. (1994). Coffee Creek Landcare project zoological study: a resource for the monitoring and maintenance of wildlife. Report to the Huntingfield/Coffee Creek Landcare Group, 1 September 1994.

Heddle, E.M. and Specht, R.L. (1975). Dark Island heath (Ninety-Mile Plain, South Australia) VIII. The effect of fertilizers on composition and growth, 1950–1972. *Australian Journal of Botany* **23**: 151–164.

Heil, G.W. and Diemont, W.H. (1983). Raised nutrient levels change heathland into grassland. *Vegetatio* **53**: 113–120.

Hester, A.J. and Hobbs, R.J. (1992). Influence of fire and soil nutrients on native and non-native annuals at remnant vegetation edges in the Western Australian wheatbelt. *Journal of Vegetation Science* **3**: 101–108.

Hobbs, R.J. (1987). Disturbance regimes in remnants of natural vegetation. Eds Saunders, D.A., Arnold, G.W., Burbidge, A.A. and Hopkins, A.J.M., *Nature Conservation: The Role*

of Remnants of Native Vegetation, pp 233–240. Surrey Beatty and Sons, Chipping Norton, NSW.

Hobbs, R.J. (1989). The nature and effects of disturbance relative to invasions. Eds Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmánek, M. and Williamson, M., *Biological Invasions: A Global Perspective*, pp 389–405. John Wiley and Sons, Chichester.

Hobbs, R.J. (1991). Disturbance as a precursor to weed invasion in native vegetation. *Plant Protection Quarterly* **6**(3): 99–104.

Hobbs, R. J. (2001). Synergisms among habitat fragmentation, livestock grazing, and biotic invasions in southwestern Australia. *Conservation Biology* **15**: 1522–1528.

Hobbs, R.J. and Atkins, L. (1988). Effect of disturbance and nutrient addition on native and introduced annuals in plant communities in the Western Australian wheatbelt. *Australian Journal of Ecology* **13**: 171–179.

Hobbs, R.J. and Atkins, L. (1991). Interactions between annuals and woody perennials in a Western Australian nature reserve. *Journal of Vegetation Science* **2**: 643–654.

Hobbs, R.J., Saunders, D.A. and Main, A.R. (1993). Conservation management in fragmented systems. Eds Hobbs, R.J. and Saunders, D.A., *Reintegrating Fragmented Landscapes*, pp 279–296. Springer-Verlag, New York.

Holmes, P.M. (2002). Depth distribution and composition of seed-banks in alien-invaded and uninvaded fynbos vegetation. *Austral Ecology* **27**: 110–120.

Hopmans, P., Stewart, H.T.L., Flinn, D.W. and Hillman, T.J. (1990). Growth, biomass production and nutrient uptake by seven tree species irrigated with municipal effluent at Wodonga, Australia. *Forest Ecology and Management* **30**: 203–211.

Hopper, S.D. and Muir, B.G. (1984). Conservation of the Kwongan. Eds Pate, J.S. and Beard, J.S., *Kwongan: Plant Life of the Sandplain*, pp 253–266. University of Western Australia Press, Nedlands.

Jackson, M.L. (1958). *Soil Chemical Analysis*. Prentice-Hall, Englewood Cliffs, N.J.

Janzen, D.H. (1983). No park is an island: increase in interference from outside as park size decreases. *Oikos* **41**: 402–410.

Johnson, W.W. and Sanders, H.O. (1977). Chemical Forest Fire Retardants: Acute Toxicity to Five Freshwater Fishes and a Scud. Technical Paper, 1977. United States Department of the Interior, Fish and Wildlife Service, Washington, DC.

Jones, R. (1968). Productivity studies on heath vegetation in southern Australia. The use of fertilizer in studies of production processes. *Folia Goebotanica Phytotax. (Praha)* **3**: 355–362.

Jones, R., Groves, R.H. and Specht, R.L. (1969). Growth of heath vegetation III. Growth curves for heaths in southern Australia: a reassessment. *Australian Journal of Botany* **17**: 309–314.

Kapos, V. (1989). Effects of isolation on the water status of forest patches in the Brazilian Amazon. *Journal of Tropical Ecology* **5**: 173–185.

Keeley, S.C., Keeley, J.E., Hutchinson, S.M. and Johnson, A.W. (1981). Postfire succession of the herbaceous flora in southern California chaparral. *Ecology* **62**(6): 1608–1621.

Keith, D.A. and Bradstock, R.A. (1994). Fire and competition in Australian heath: a conceptual model and field investigations. *Journal of Vegetation Science* **5**: 347–354.

Keith, D.A. and Myerscough, P.J. (1993). Floristics and soil relations of upland swamp vegetation near Sydney. *Australian Journal of Ecology* **18**: 325–344.

Kirkpatrick, J.B. (1977). *The Disappearing Heath.* Tasmanian Conservation Trust, Hobart.

Kirkpatrick, J.B. (1999). The characteristics and management problems of the vegetation and flora of the Huntingfield area, southern Tasmania. *Papers and Proceedings of the Royal Society of Tasmania* **133**(1): 103–113.

Kirkpatrick, J.B. and Dickinson, K.J.M. (1982). Recent destruction of natural vegetation in Tasmania. *Search* **13**(7–8): 186–187.

Kirkpatrick, J.B. and Gilfedder, L. (1995). Maintaining integrity compared with maintaining rare and threatened taxa in remnant bushland in subhumid Tasmania. *Biological Conservation* **74**: 1–8.

Kirkpatrick, J.B. and Harris, S. (1999a). Coastal, heath and wetland vegetation. Eds Reid, J.B., Hill, R.S., Brown, M.J. and Hovenden, M.J., *Vegetation of Tasmania.* Flora of Australia Supplementary Series No. 8, pp 304–332. Australian Biological Resources Study, Department of the Environment and Heritage, Hobart.

Kirkpatrick, J.B. and Harris, S. (1999b). *The Disappearing Heath Revisited.* Tasmanian Environment Centre Inc., Hobart.

Kirkpatrick, J.B. and McQuillan, P.B. (1996). Biological resources, reservation and management of the Huntingfield Estate, Tasmania. Unitas Pty Ltd, Report to the Huntingfield/Coffee Creek Landcare Group, November 1996.

Kirkpatrick, J.B., Barker, P., Brown, M.J., Harris, S. and Mackie, R. (1995). The reservation status of Tasmanian vascular plant communities. Parks and Wildlife Service, Tasmania, Wildlife Scientific Report, No. 95/4.

Kruger, F.J. (1981). Conservation: South African heathlands. Ed. Specht, R.L., *Heathlands and Related Shrublands: Analytical Studies.* Ecosystems of the World, Vol. 9B, pp 231–234. Elsevier, Amsterdam.

Kruger, F.J. (1983). Plant community diversity and dynamics in relation to fire. Eds Kruger, F.J., Mitchell, D.T. and Jarvis, J.U.M., *Mediterranean-Type Ecosystems: The Role of Nutrients,* pp 446–472. Springer-Verlag, Heidelberg.

Kruger, F.J., Breytenbach, G.J., Macdonald, I.A.W. and Richardson, D.M. (1989). The characteristics of invaded Mediterranean-climate regions. Eds Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmánek, M. and Williamson, M., *Biological Invasions: A Global Perspective.* SCOPE, Vol. 37, pp 181–213. John Wiley and Sons, Chichester.

Lambert, M.J. and Turner, J. (1987). Suburban development and change in vegetation nutritional status. *Australian Journal of Ecology* **12**: 193–196.

Lamont, B.B., Swanborough, P.W. and Ward, D. (2000). Plant size and season of burn affect flowering and fruiting of the grasstree *Xanthorrhoea preissii. Austral Ecology* **25**(3): 268–272.

Landsberg, J., Morse, J. and Khanna, P. (1990). Tree dieback and insect damage in remnants of native woodlands on farms. *Proceedings of the Ecological Society of Australia* **16**: 149–165.

Lane, P., Morris, D. and Shannon, G. (1999). *Common Grasses of Tasmania, an Agriculturalist's Guide.* Tasmanian Environment Centre, Hobart.

Larson, D.L. and Newton, W.E. (1996). Effects of fire retardant chemicals and fire suppressant foam on North Dakota prairie vegetation. *Proceedings of the North Dakota Academy of Science* **50**: 137–144. Northern Prairie Wildlife Research Center Home Page, http://www.npwrc.usgs.gov/resource/othrdata/fireweb/larson/larson.htm#methods. USA, version 02MAR98.

Larson, J.R. and Duncan, D.A. (1982). Annual grassland response to fire retardant and wildfire. *Journal of Rangeland Management* **35**: 700–703.

Laurance, W.F. (1991). Edge effects in tropical forest fragments: application of a model for the design of nature reserves. *Biological Conservation* **57**: 205–219.

Laurance, W.F. (2000). Do edge effects occur over large spatial scales? *Trends in Ecology and Evolution* **15**: 134–135.

Laurance, W. F., Lovejoy, T. E., Vasconcelos, H. L., Bruna, E. M., Didham, R. K., Stouffer, P. C., Gascon, C., Bierregaard, R. O., Laurance, S. G. and Sampaio, E. (2002). Ecosystem decay of Amazonian forest fragments: a 22-year investigation. *Conservation Biology* **16**: 605–618.

Le Brocque, A.F. and Buckney, R.T. (1995). Vegetation and environmental patterns on soils derived from Hawkesbury Sandstone and Narrabeen substrata in Ku-ring-gai Chase National Park, New South Wales. *Australian Journal of Ecology* **20**: 229–238.

Lehane, R. (1995). Effluent irrigated plantations: design and management. CSIRO Division of Forestry, Technical Paper No. 2.

Leigh, J.H. and Holgate, M.D. (1979). The responses of the understorey of forests and woodlands of the Southern Tablelands to grazing and burning. *Australian Journal of Ecology* **4**: 25–45.

Leigh, J.H., Wimbush, D.J., Wood, D.H., Holgate, M.D., Slee, A.V., Stanger, M.G. and Forrester, R.I. (1987). Effects of rabbit grazing and fire on a subalpine environment. I. herbaceous and shrubby vegetation. *Australian Journal of Botany* **35**: 433–464.

Lepart, J. and Debussche, M. (1991). Invasion processes as related to succession and disturbance. Eds Groves, R.H. and di Castri, F., *Biogeography of Mediterranean Invasions*, pp 159–177. Cambridge University Press, Cambridge.

Lewis, C.E. and Harshbarger, T.J. (1976). Shrub and herbaceous vegetation after 20 years of prescribed burning in the South Carolina coastal plain. *Journal of Rangeland Management* **29**(1): 13–18.

Lidicker, W.Z. (1999). Responses of mammals to habitat edges: an overview. *Landscape Ecology* **14**(4): 333–343.

Loneragan, O.W. and Loneragan, J.F. (1964). Ashbed and nutrients in the growth of seedlings of karri. *Journal of the Royal Society of Western Australia* **47**: 74–80.

Lovejoy, T.E., Bierregaard Jr, B.O., Rylands, A.B., Malcolm, J.R., Quintela, C.E., Harper, L.H., Brown Jr, K.S., Powell, A.H., Powell, G.V.N., Schubart, H.O.R. and Hays, M.B. (1986). Edge and other effects of isolation on Amazon forest fragments. Ed. Soulé, M.E., *Conservation Biology. The Science of Scarcity and Diversity*, pp 257–285. Sinauer Associates, Sunderland, Massachusetts.

Low, A.B. and Lamont, B.B. (1985). Nutrient allocation in native heath and rehabilitated communities. Eds Lamont, B. and Low, A.B., *Proceedings, Plant Ecology of the Eneabba Heathlands*, Perth, West Australian Institute of Technology.

Low, A.B. and Lamont, B.B. (1986). Nutrient allocation in winter rainfall proteaceous heathlands in relation to nutrient losses through wildflower picking and fire. *Acta Horticulturae* **185**: 89–99.

Low, A.B. and Lamont, B.B. (1990). Aerial and below-ground phytomass of *Banksia* scrub-heath at Eneabba, south-western Australia. *Australian Journal of Botany* **38**: 351–359.

MacArthur, R.H. and Wilson, E.O. (1967). *The Theory of Island Biogeography*. Princeton University Press, Princeton, New Jersey, USA.

MacDonald, I.A.W., Graber, D.M., DeBenedetti, S., Groves, R.H. and Fuentes, E.R. (1988). Introduced species in nature reserves in Mediterranean-type climate regions of the world. *Biological Conservation* **44**: 37–66.

Malajczuk, N. and Glenn, A.R. (1981). *Phytophthora cinnamomi* — a threat to the heathlands. Ed. Specht, R.L., *Heathlands and Related Shrublands: Analytical Studies*. Ecosystems of the World, Vol. 9B, pp 241–247. Elsevier, Amsterdam.

Mallik, A.U. and Gimingham, C.H. (1983). Regeneration of heathland plants following burning. *Vegetatio* **53**: 45–58.

Margaris, N.S., Adamandiadou, S., Siafaca, L. and Diamantopoulos, J. (1984). Nitrogen and phosphorus content in plant species of Mediterranean ecosystems in Greece. *Vegetatio* **55**: 29–35.

Marsden-Smedley, J. (1992). Application of fire suppression foam in buttongrass moorlands. Department of Parks, Wildlife and Heritage, Hobart, Memo, M1/25/1(6), 4 March 1992.

Martin, T.G. and Catterall, C.P. (2001). Do fragmented coastal heathlands have habitat value to birds in eastern Australia? *Wildlife Research* **28**: 17–31.

Matlack, G.R. (1993). Microenvironment variation within and among forest edge sites in the eastern United States. *Biological Conservation* **66**(3): 185–194.

Maze, K.E. and Bond, W.J. (1996). Are *Protea* populations seed limited? Implications for wildflower harvesting in Cape fynbos. *Australian Journal of Ecology* **21**(1): 96–105.

McCune, B. and Mefford, M.J. (1999). *PC-ORD. Multivariate Analysis of Ecological Data. Version 4*. MjM Software Design, Gleneden Beach, Oregon, USA.

McDonald, R.C., Isbell, R.F., Speight, J.G., Walker, J. and Hopkins, M.S. (1984). *Australian Soil and Land Survey Field Handbook*. Inkata Press, Melbourne.

McDonald, S.F., Hamilton, S.J., Buhl, K.J. and Heisinger, J.F. (1996). Acute toxicity of fire control chemicals to *Daphnia magna* (Straus) and *Selenastrum capricornutum* (Printz). *Ecotoxicology and Environmental Safety* **33**: 62–72.

McDonald, S.F., Hamilton, S.J., Buhl, K.J. and Heisinger, J.F. (1997). Acute toxicity of fire-retardant and foam-suppressant chemicals to *Hyalella azteca* (Saussure). *Environmental Toxicology and Chemistry* **16**(7): 1370–1376.

McFarland, D.C. (1990). Flower and seed phenology of some plants in the subtropical heathlands of Cooloola National Park, Queensland, Australia. *Australian Journal of Botany* **38**: 501–509.

McLendon, T. and Redente, E.F. (1991). Nitrogen and phosphorus effects on secondary succession dynamics on a semi-arid sagebrush site. *Ecology* **72**: 2016–2024.

McMaster, G.S., Jow, W.M. and Kummerow, J. (1982). Response of *Adenostoma fasciculatum* and *Ceanothus greggii* chaparral to nutrient additions. *Journal of Ecology* **70**: 745–765.

Mendoza, R.E., Collantes, M.B., Anchorena, J. and Cainzos, S. (1995). Effects of liming and fertilization on forage yield and vegetation in dry heath soil from Tierra del Fuego. *Journal of Plant Nutrition* **18**(3): 401–420.

Milberg, P. and Lamont, B.B. (1995). Fire enhances weed invasion of roadside vegetation in southwestern Australia. *Biological Conservation* **73**: 45–59.

Miller, R.W. and Donahue, R.L. (1995). *Soils in Our Environment*. Prentice Hall, New Jersey.

Minchin, P.R. (1987). An evaluation of the relative robustness of techniques for ecological ordination. *Vegetatio* **69**: 89–107.

Molnar, C.D., Fletcher, D. and Parsons, R.F. (1989). Relationships between heath and *Leptospermum laevigatum* scrub at Sandringham, Victoria. *Proceedings of the Royal Society of Victoria* **101**: 77–87.

Moodie, E.G., Stewart, R.S. and Bowen, S.E. (1986). The impact of surfactants on Norfolk Island Pines along Sydney coastal beaches since 1973. *Environmental Pollution (Series A)* **41**: 153–164.

Moro, D. (1991). The distribution of small mammal species in relation to heath vegetation near Cape Otway, Victoria. *Wildlife Research* **18**(5): 605–618.

Morrison, A.B. and Purling, T.J. (1982). A report to the Western Australian Department of Fisheries and Wildlife covering observations on Cape Barren Geese held at the Poultry Research Station, Woodlands, W.A. from mid 1978 until April 1982. Poultry Research Station, Woodlands, W.A.

Morrison, D.A., Buckney, R.T., Bewick, B.J. and Cary, G.J. (1996). Conservation conflicts over burning bush in south-eastern Australia. *Biological Conservation* **76**: 167–175.

Morrison, D.A., Cary, G.J., Pengelly, S.M., Ross, D.G., Mullins, B.J., Thomas, C.R. and Anderson, T.S. (1995). Effects of fire frequency on plant species composition of sandstone communities in the Sydney region: Inter-fire interval and time-since-fire. *Australian Journal of Ecology* **20**(2): 239–247.

Mueller-Dombois, D. and Ellenberg, H. (1974). *Aims and Methods of Vegetation Science.* Wiley and Sons, New York.

Muir, B.G. (1979). Observations on wind-blown superphosphate in native vegetation. *Western Australian Naturalist* **14**: 128–130.

Munsell Soil Color Company (1954). *Revised Standard Soil Color Charts.* Munsell Color Company Inc., Baltimore.

Murcia, C. (1995). Edge effects in fragmented forests — implications for conservation. *Trends in Ecology and Evolution* **10**(2): 58–62.

Myers, B.J. (1992). Effluent loading rates for irrigated plantations — a water balance model. *Australian Forestry* **55**: 39–47.

Myers, B.J., Theiveyanathan, S., O'Brien, N.D. and Bond, W.J. (1996). Growth and water use of effluent-irrigated *Eucalyptus grandis* and *Pinus radiata* plantations. *Tree Physiology* **16**: 211–219.

Myerscough, P.J. and Carolin, R.C. (1986). The vegetation of the Eurunderee sand mass, headlands and previous islands in the Myall Lakes area, New South Wales. *Cunninghamia* **1**(4): 399–466.

Myerscough, P.J., Clarke, P.J. and Skelton, N.J. (1995). Plant coexistence in coastal heaths: floristic patterns and species attributes. *Australian Journal of Ecology* **20**: 482–493.

Naiman, R.J. and Décamps, H. (1991). Landscape boundaries in the management and restoration of changing environments: A summary. Eds Holland, M.M., Risser, P.G. and Naiman, R.J., *Ecotones: The Role of Landscape Boundaries in the Management and Restoration of Changing Environments,* pp 130–137. Chapman and Hall, New York.

Nieuwenhuis, A. (1987). The effect of fire frequency on the sclerophyll vegetation of the West Head, New South Wales. *Australian Journal of Ecology* **12**: 373–385.

Norecol (1989). Toxicological review of fire fighting foams. Report prepared for Fire Management Forest Protection Branch, Ministry of Forests and Lands, Victoria, British Columbia, by Norecol Environmental Consultants Ltd, Vancouver, British Columbia, Report no. 1-091-02.01, April 1989.

Odum, E.P. (1959). *Fundamentals of Ecology.* W B Saunders Company, Philadelphia, USA.

Ogle, C.C. (1987). The incidence and conservation of animal and plant species in remnants of native vegetation within New Zealand. Eds Saunders, D.A., Arnold, G.W., Burbidge, A.A. and Hopkins, A.J.M., *Nature Conservation: The Role of Remnants of Native Vegetation,* pp 79–87. Surrey Beatty and Sons, Chipping Norton, NSW.

Olesen, S.E. and Mark, H.S. (1991). Long-term effects of sewage sludge application in a conifer plantation on a sandy soil. Ed. Hall, J.E., *Alternative uses for sewage sludge. Proceedings of the WRc International Workshop*, University of York, UK, 5–7 September 1989. Pergamon Press, Oxford.

Ozanne, P.G. and Specht, R.L. (1981). Mineral nutrition of heathlands: phosphorus toxicity. Ed. Specht, R.L., *Heathlands and Related Shrublands: Analytical Studies.* Ecosystems of the World, Vol. 9B, pp 209–213. Elsevier, Amsterdam.

Ozanne, P.G., Kirton, D.J. and Shaw, T.C. (1961). The loss of phosphorus from sandy soils. *Australian Journal of Agricultural Research* **12**: 409–423.

Palik, B.J. and Murphy, P.G. (1990). Disturbance versus edge effects in sugar-maple/beech forest fragments. *Forest Ecology and Management* **32**: 187–202.

Parks and Wildlife Service Tasmania (1997). Peter Murrell Nature Reserve and Conservation Area Interim Management Strategy. Parks and Wildlife Tasmania, Department of Environment and Land Management, Hobart, August 1997.

Pate, J.S. and Dell, B. (1982). Economy of mineral nutrients in sandplain species. Eds Pate, J.S. and Beard, J.S., *Kwongan: Plant Life of the Sandplain*, pp 227–252. University of Western Australia Press, Nedlands.

Patric, J.H. and Hanes, T.L. (1964). Chaparral succession in a San Gabriel mountain area of California. *Ecology* **45**: 353–360.

Pelton, G.A. and Conran, J.G. (2002). Comparison of two rolled sandy heath communities within a single fire patch in Ngarkat Conservation Park, South Australia. *Austral Ecology* **27**: 85–93.

Pettit, N.E., Froend, R.H. and Ladd, P.G. (1995). Grazing in remnant woodland vegetation: changes in species composition and life form groups. *Journal of Vegetation Science* **6**: 121–130.

Podger, F.D. (1989). Comparative pathogenicity of fourteen Australian isolates of *Phytophthora cinnamomi* determined on transplants of Tasmanian temperate heathland. *Australian Journal of Botany* **37**: 491–500.

Podger, F. D. (1990). *Phytophthora*, fire and change in vegetation. *Tasforests* **2**(2): 125–128.8

Podger, F.D. and Brown, M.J. (1989). Vegetation damage caused by *Phytophthora cinnamomi* on disturbed sites in temperate rainforest in western Tasmania. *Australian Journal of Botany* **37**: 443–480.

Podger, F.D., Mummery, D.C., Palzer, C.R. and Brown, M.J. (1990a). Bioclimatic analysis of the distribution of damage to native plants in Tasmania by *Phytophthora cinnamomi*. *Australian Journal of Ecology* **15**: 281–289.

Podger, F.D., Palzer, C. and Wardlaw, T. (1990b). A guide to the Tasmanian distribution of *Phytophthora cinnamomi* and its effects on native vegetation. *Tasforests* **2**: 13–20.

Posamentier, H.G., Clark, S.S., Hain, D.L. and Recher, H.F. (1981). Succession following wildfire in coastal heathland (Nadgee Nature Reserve NSW). *Australian Journal of Ecology* **6**: 165–175.

Posse, G. and Mendoza, R.E. (1995). Yield response curves of *Dactylis glomerata* to addition of P, N, K and lime on a xeric soil of Tierra del Fuego. *Communications in Soil Science and Plant Analysis* **26**(3–4): 631–642.

Pratt, B.H. and Heather, W.A. (1973). The origin and distribution of *Phytophthora cinnamomi* Rands in Australian native plant communities and the significance of its association with particular plant species. *Australian Journal of Biological Science* **26**: 559–573.

Purdie, R.W. (1977a). Early stages of regeneration after burning in dry sclerophyll vegetation I. Regeneration of the understorey by vegetative means. *Australian Journal of Botany* **25**: 21–34.

Purdie, R.W. (1977b). Early stages of regeneration after burning in dry sclerophyll vegetation II. Regeneration by seed germination. *Australian Journal of Botany* **25**: 35–46.

Pywell, R.F., Webb, N.R. and Putwain, P.D. (1994). Soil fertility and its implications for the restoration of heathland on farmland in southern Britain. *Biological Conservation* **70**(2): 169–181.

Ranney, J.W., Bruner, M.C. and Levenson, J.B. (1981). The importance of edge in the structure and dynamics of forest islands. Eds Burgess, R.L. and Sharpe, D.M., *Forest Island Dynamics in Man-Dominated Landscapes*. Ecological Studies, Vol. 41, pp 67–95. Springer-Verlag, New York.

Rawet, D., Smith, R. and Kravainis, G. (1996). A comparison of water additives for mopping-up after forest fires. *International Journal of Wildland Fire* **6**(1): 37–43.

Raymont, G.E. and Higginson, F.R. (1992). *Australian Laboratory Handbook of Soil and Water Chemical Methods*. Inkata Press, Melbourne.

Rejmánek, M. (1989). Invasibility of plant communities. Eds Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Rejmánek, M. and Williamson, M., *Biological Invasions: a Global Perspective*. SCOPE, pp 369–388. John Wiley and Sons, Chichester.

Rose, R.J. and Webb, N.R. (1994). The effects of temporary ballast roadways on heath vegetation. *Journal of Applied Ecology* **31**(4): 642–650.

Rose, S. (1997). Influence of suburban edges on invasion of *Pittosporum undulatum* into the bushland of northern Sydney, Australia. *Australian Journal of Ecology* **22**(1): 89–100.

Rundel, P.W. (1983). Impact of fire on nutrient-cycles in Mediterranean-type ecosystems with reference to chaparral. Eds Kruger, F.J., Mitchell, D.T. and Jarvis, J.U.M., *Mediterranean-Type Ecosystems: The Role of Nutrients*, pp 192–207. Springer-Verlag, Heidelberg.

Russell, R.P. and Parsons, R.F. (1978). Effects of time since fire on heath floristics at Wilsons Promontory, southern Australia. *Australian Journal of Botany* **26**: 53–61.

Saunders, D.A., Arnold, G.W., Burbidge, A.A. and Hopkins, A.J.M. (1987). The role of remnants of native vegetation in nature conservation: future directions. Eds Saunders, D.A., Arnold, G.W., Burbidge, A.A. and Hopkins, A.J.M., *Nature Conservation: the Role of Remnants of Native Vegetation*, pp 387–392. Surrey Beatty and Sons, Chipping Norton, NSW.

Saunders, D.A., Hobbs, R.J. and Margules, C.R. (1991). Biological consequences of ecosystem fragmentation: a review. *Conservation Biology* **5**(1): 18–32.

Schaefer, T.H. (1996). Environmental impact of Class A fire fighting foams. 3M Australia Pty Ltd, Paper presented to TFS Class A foam workshop, Report no. 009333.

Schlobohm, P. (1993). Understanding the environmental effects of class A foams. *Foam Applications for Wildland and Urban Fire Management* **5**(2): 26–30.

Scougall, S.A., Majer, J.D. and Hobbs, R.J. (1993). Edge effects in grazed and ungrazed Western Australian wheatbelt remnants in relation to ecosystem reconstruction. Eds Saunders, D.A., Hobbs, R.J. and Ehrlich, P.R., *Nature Conservation 3: Reconstruction of Fragmented Ecosystems*, pp 163–178. Surrey Beatty and Sons, Chipping Norton, NSW.

Shearer, B.L. and Tippett, J.T. (1989). Jarrah dieback: the dynamics and management of *Phytophthora cinnamomi* in the Jarrah (*Eucalyptus marginata*) forest of south-western Australia. Department of Conservation and Land Management, Western Australia, Research Bulletin, No. 3, November 1989.

Siddiqi, M.Y. and Carolin, R.C. (1976). Studies in the ecology of coastal heath in New South Wales II. The effects of water supply and phosphorus uptake on the growth of *Banksia serratifolia, B. aspleniifolia* and *B. ericifolia. Proceedings of the Linnean Society of New South Wales* **101**(1): 38–52.

Siddiqi, M.Y., Carolin, R.C. and Myerscough, P.J. (1976a). Studies in the ecology of coastal heath in New South Wales III. Regrowth of vegetation after fire. *Proceedings of the Linnean Society of New South Wales* **101**(1): 53–62.

Siddiqi, M.Y., Myerscough, P.J. and Carolin, R.C. (1976b). Studies in the ecology of coastal heath in New South Wales IV. Seed survival, germination, seedling establishment and early growth in *Banksia serratifolia* Salisb., *B. aspleniifolia* Salisb. and *B. ericifolia* L.F. in relation to fire: temperature and nutritional effects. *Australian Journal of Ecology* **1**: 175–183.

Sinclaire Knight Mertz (1996). Steiglitz sewerage treatment plant irrigation scheme: environmental management plan. Break O'Day Council, Draft unpublished report, No. TA 15095, 19/9/96.

Sisk, T.D. and Margules, C.R. (1993). Habitat edges and restoration: methods for quantifying edge effects and predicting the results of restoration efforts. Eds Saunders, D.A., Hobbs, R.J. and Ehrlich, P.R., *Nature Conservation 3: Reconstruction of Fragmented Ecosystems*, pp 57–69. Surrey Beatty and Sons, Chipping Norton, NSW.

Sloane, D.J. and Weldon, B.D. (1996). Groundwater and soil assessment: Parnella effluent irrigation area, St Helens. Sloane Weldon Pty Ltd, Engineering and Groundwater Geologists, Client: Sinclair Knight Merz, Report no. 1996/107, 12 December 1996.

Sloane, D.J. and Weldon, B.D. (1997). Geotechnical assessment: proposed sewage treatment lagoon, Stieglitz, St Helens. Sloane Weldon Pty Ltd, Engineering and Groundwater Geologists, Client: Sinclair Knight Merz, Report no. 1997/08, 17 February 1997.

Sokal, R.R. and Rohlf, F.J. (1995). *Biometry*. W H Freemon, New York.

Soulé, M.E., Alberts, A.C. and Bolger, D.T. (1992). The effects of habitat fragmentation on chaparral plants and vertebrates. *Oikos* **63**: 39–47.

Specht, R.L. (1963). Dark Island heath (Ninety-Mile Plain, South Australia) VII. The effect of fertilizers on composition and growth, 1950–60. *Australian Journal of Botany* **11**: 67–94.

Specht, R.L. (1966). The growth and distribution of mallee-broombush (*Eucalyptus incrassata – Melaleuca uncinata* association) and heath vegetation near Dark Island Soak, Ninety-Mile Plain, South Australia. *Australian Journal of Botany* **14**: 361–371.

Specht, R.L. (1969a). A comparison of the sclerophyllous vegetation characteristics of Mediterranean-type climates in France, California and Australia I. Structure, morphology and succession. *Australian Journal of Botany* **17**: 277–292.

Specht, R.L. (1969b). A comparison of the sclerophyllous vegetation characteristics of Mediterranean-type climates in France, California and Australia II. Dry matter, energy and nutrient accumulation. *Australian Journal of Botany* **17**: 293–308.

Specht, R.L. (1975). The effect of fertilizers on sclerophyll (heath) vegetation — the problems of revegetation after sand-mining of high dunes. *Search* **6**(11–12): 459–461.

Specht, R.L. (1976). Heath soils and phosphorus. *Search* **7**(4): 112–113.

Specht, R.L. (1981a). Conservation: Australian heathlands. Ed. Specht, R.L., *Heathlands and Related Shrublands: Analytical Studies*. Ecosystems of the World, Vol. 9B, pp 235–240. Elsevier, Amsterdam.

Specht, R.L. (1981b). Heathlands and related shrublands of the world. Ed. Specht, R.L., *Heathlands and Related Shrublands: Descriptive Studies*. Ecosystems of the World, Vol. 9A, pp 1–18. Elsevier, Amsterdam.

Specht, R.L. (1981c). Nutrient release from decomposing leaf litter of *Banksia ornata*, Dark Island heathland, South Australia. *Australian Journal of Ecology* **6**: 59–63.

Specht, R.L. (1981d). The sclerophyllous (heath) vegetation of Australia: the eastern and central states. Ed. Specht, R.L., *Heathlands and Related Shrublands: Descriptive Studies*. Ecosystems of the World, Vol. 9A, pp 125–210. Elsevier, Amsterdam.

Specht, R.L. and Cleland, J.B. (1961). Flora conservation in South Australia I. The preservation of plant formations and associations recorded in South Australia. *Transactions of the Royal Society of South Australia* **85**: 177–196.

Specht, R.L. and Groves, R.H. (1966). A comparison of the phosphorus nutrition of Australian heath plants and introduced economic plants. *Australian Journal of Botany* **14**: 201–221.

Specht, R.L. and Moll, E.J. (1983). Mediterranean-type heathlands and sclerophyllous shrublands of the world: an overview. Eds Kruger, F.J., Mitchell, D.T. and Jarvis, J.U.M., *Mediterranean-Type Ecosystems: The Role of Nutrients*, pp 41–65. Springer-Verlag, Heidelberg.

Specht, R.L. and Rayson, P. (1957). Dark Island heath (Ninety-Mile Plain, South Australia) I. Definition of the ecosystem. *Australian Journal of Botany* **5**: 52–85.

Specht, R.L. and Specht, A. (1989). Species richness of sclerophyll (heathy) plant communities in Australia — the influence of overstorey cover. *Australian Journal of Botany* **37**: 337–350.

Specht, R.L., Connor, D.J. and Clifford, H.T. (1977). The heath–savannah problem: the effect of fertilizer on sand-heath vegetation of North Stradbroke Island, Queensland. *Australian Journal of Ecology* **2**: 179–186.

Specht, R.L., Rayson, P. and Jackman, M.E. (1958). Dark Island heath (Ninety-Mile Plain, South Australia) VI. Pyric succession: changes in composition, coverage, dry weight, and mineral nutrient status. *Australian Journal of Botany* **6**: 59–92.

Start, A.N. (1991). How can edge effects be minimized? Eds Saunders, D.A. and Hobbs, R.J., *Nature Conservation 2: the Role of Corridors*, pp 417–418. Surrey Beatty and Sons, Chipping Norton, NSW.

Stewart, H.T.L. and Flinn, D.W. (1984). Establishment and early growth of trees irrigated with wastewater at four sites in Victoria, Australia. *Forest Ecology and Management* **8**: 243–256.

Stewart, H.T.L., Hopmans, P., Flinn, D.W. and Hillman, T.J. (1990). Nutrient accumulation in trees and soil following irrigation with municipal effluent in Australia. *Environmental Pollution* **63**: 155–177.

Tamm, C.O. and Troedsson, T. (1955). An example of amounts of plant nutrients supplied to ground in road dust. *Oikos* **6**: 61–70.

Thatcher, A.C. and Westman, W.E. (1975). Succession following mining on high dunes of coastal southeast Queensland. *Proceedings of the Ecological Society of Australia* **9**: 17–33.

Thomas, J.F., Gomboso, J., Oliver, J.E. and Ritchie, V.A. (1997). Wastewater re-use, stormwater management, and the National Water Reform Agenda. CSIRO Land and Water, Research Position Paper 1, 1997.

USDA (1988). Fire fighting chemical products — differences and similarities. United States Department of Agriculture, http://www.fs.fed.us/rm/fire/Fire_Chemicals_Defined.html. USA.

Veitch, N., Webb, N.R. and Wyatt, B.K. (1995). The application of geographic information systems and remotely sensed data to the conservation of heathland fragments. *Biological Conservation* **72**: 91–97.

Vyas, N.B., Spann, J.W. and Hill, E.F. (1996). Effects of Silv-Ex on terrestrial wildlife. *Proceedings of the North Dakota Academy of Science* **50**: 150–151

Wace, N. (1977). Assessment of dispersal of plant species — car-borne flora in Canberra. *Proceedings of the Ecological Society of Australia* **10**: 167–186.

Wales, B.A. (1972). Vegetation analysis of north and south edges in a mature oak-hickory forest. *Ecological Monographs* **42**: 451–471.

Watts, S.E. (1998). Short-term influence of tank tracks on vegetation and microphytic crusts in shrubsteppe habitat. *Environmental Management* **22**(4): 611–616.

Webb, N.R. (1990). Changes on the heathlands of Dorset, England, between 1978 and 1987. *Biological Conservation* **51**: 273–286.

Webb, N.R. and Vermaat, A.H. (1990). Changes in vegetational diversity on remnant heathland fragments. *Biological Conservation* **53**(4): 253–264.

Weste, G. (1974). *Phytophthora cinnamomi*: the cause of severe disease in certain native communities in Victoria. *Australian Journal of Botany* **22**: 1–8.

Weste, G. and Kennedy, J. (1997). Regeneration of susceptible species following a decline of *Phytophthora cinnamomi* over a period of 20 years on defined plots in the Grampians, western Victoria. *Australian Journal of Botany* **45**(1): 167–190.

Weste, G. and Law, C. (1973). The invasion of native forests by *Phytophthora cinnamomi* III. Threat to the National Park, Wilsons Promontory, Victoria. *Australian Journal of Botany* **21**: 31–51.

Wiens, J.A., Crawford, C.S. and Gosz, J.R. (1985). Boundary dynamics: a conceptual framework for studying landscape ecosystems. *Oikos* **45**: 421–427.

Wilcove, D.S., McLellen, C.J. and Dobson, A.P. (1986). Habitat fragmentation in the temperate zone. Ed. Soulé, M.E., *Conservation Biology, the Science of Scarcity and Diversity*, pp 237–256. Sinauer, Sunderland, Massachusetts.

Williams, K. (1991). Dry sclerophyll vegetation. Ed. Kirkpatrick, J.B., *Tasmanian Native Bush: A Management Handbook*, pp 54–75. Tasmanian Environment Centre Inc., Hobart.

Williams, R.J. (1992). Gap dynamics in subalpine heathland and grassland vegetation in south-eastern Australia. *Australian Journal of Ecology* **80**: 343–352.

Wills, R. T. (1993). The ecological impact of *Phytophthora cinnamomi* in the Stirling Range National Park. *Australian Journal of Ecology* **18**: 145–159.

Wilson, A.D. (1990). The effect of grazing on Australian ecosystems. *Proceedings of the Ecological Society of Australia* **16**: 235–244.

Witkowski, E.T.F. and Mitchell, D.T. (1987). Variations in soil phosphorus in the fynbos biome, South Africa. *Journal of Ecology* **75**: 1159–1171.

Witkowski, E.T.F., Mitchell, D.T. and Stock, W.D. (1990a). Response of a Cape fynbos ecosystem to nutrient additions: nutrient dynamics in fertilized soils. *Acta Oecologia* **11**(2): 165–179.

Witkowski, E.T.F., Mitchell, D.T. and Stock, W.D. (1990b). Response of a Cape fynbos ecosystem to nutrient additions: shoot growth and nutrient content of a proteoid (*Leucospermum parile*) and an ericoid (*Phylica cephalantha*) evergreen shrub. *Acta Oecologia* **11**(3): 311–326.

Yates, C.J., Norton, D.A. and Hobbs, R.J. (2000). Grazing effects on plant cover, soil and microclimate in fragmented woodlands in south-western Australia: implications for restoration. *Austral Ecology* **25**(1): 36–47.

Young, A. and Mitchell, N. (1994). Microclimate and vegetation edge effects in a fragmented podocarp-broadleaf forest in New Zealand. *Biological Conservation* **67**(1): 63–72.

Zar, J.H. (1996). *Biostatistical Analysis*. Prentice Hall, New Jersey.

Ziegeler, D. (1994). *Prasophyllum concinnum* flora recovery plan: management phase. Parks and Wildlife Service, Tasmania, Wildlife Report, No. 94/11, 1994.

8. Appendices

APPENDIX 1. SPECIES LIST FOR STIEGLITZ

# denotes threatened species

SCIENTIFIC NAME	COMMON NAME
Dicotyledonae	
Apiaceae	
Xanthosia pilosa	Wooly Xanthosia
Asteraceae	
Argentipallium dealbatum	Everlasting
Euchiton collinus	Everlasting
Helichrysum scorpioides	Curling Everlasting
Casuarinaceae	
Allocasuarina littoralis	Black She-oak
Dilleniaceae	
Hibbertia acicularis	Prickly Guinea Flower
Hibbertia procumbens	Spreading Guinea Flower
Hibbertia riparia	Erect Guinea Flower
Droseraceae	
Drosera peltata ssp *auriculata*	Tall Sundew
Drosera pygmaea	Dwarf Sundew
Epacridaceae	
Brachyloma ciliatum	Ciliate Brachyloma
Epacris impressa	Common Heath
Leucopogon collinus	White Beard Heath
Leucopogon ericoides	Pink Beard Heath
Leucopogon virgatus	Common Beard Heath
Monotoca scoparia	Prickly Broom Heath
Styphelia adscendens	Golden Heath
Euphorbiaceae	
Amperea xiphoclada	Weaver's Plant, Broom Spurge
Fabaceae	
Aotus ericoides	Golden Pea
Bossiaea cinerea	Showy Bossiaea
Dillwynia glaberrima	Smooth Parrot Pea
Dillwynia sericea	Parrot Pea
Gompholobium huegelii	Bladder Pea
Goodeniaceae	
Dampiera stricta	Blue Dampiera
Haloragaceae	
Gonocarpus humilis	Raspwort
Gonocarpus tetragynus	Common Raspwort

SCIENTIFIC NAME	COMMON NAME
Lauraceae	
Cassytha glabella	Dodder Laurel
Loganiaceae	
Mitrasacme pilosa	Hairy Mitrewort
Mimosaceae	
Acacia genistifolia	Spreading Wattle
Acacia suaveolens	Sweet Wattle
Acacia terminalis	Sunshine Wattle
#Acacia ulicifolia	Juniper Wattle
Myrtaceae	
Baeckea ramosissima	Rosy Heath Myrtle, Baeckea
Calytrix tetragona	Common Fringe Myrtle
Eucalyptus amygdalina	Black Peppermint
Leptospermum scoparium	Tea-tree, Manuka
Pittosporaceae	
Rhytidosporum procumbens	Rhytidosporum
Proteaceae	
Banksia marginata	Honeysuckle, Silver Banksia
Hakea teretifolia	Dagger Hakea
Persoonia juniperina	Prickly Geebung
Rubiaceae	
Opercularia varia	Variable Stinkweed
Rutaceae	
Boronia pilosa	Fine Leaf Boronia
Correa reflexa	Native Fuschia
Stylidiaceae	
Styldium graminifolium	Trigger Plant
Thymelaeaceae	
Pimelea linifolia	Slender Rice Flower
Monocotyledonae	
Cyperaceae	
Baumea acuta	Twigg Rush
Gahnia radula	Cutting Grass
Lepidosperma concavum	Sand or Hill Sword Sedge
Lepidosperma filiforme	Common Rapier Sedge, Thread Rapier Sedge
Lepidosperma longitudinale	Pithy or Common Sword Sedge
Schoenus lepidosperma	Slender Bog Rush
Schoenus turbinatus	Bog Rush
Iridaceae	
Patersonia fragilis	Short Purple Flag Iris
Liliaceae	
Burchardia umbellata	Milkmaids
Laxmannia orientalis	

SCIENTIFIC NAME	COMMON NAME
Orchidaceae	
Caladenia fuscata	Dusky Caladenia
Eriochilus cucullatus	Parson's Bands or Pink Autumn Orchid
Glossodia major	Waxlip Orchid
Pterostylis spp.	Greenhood species
Thelymitra ixioides	Spotted Sun Orchid
Poaceae	
Austrodanthonia spp.	Wallaby Grass
Deyeuxia quadriseta	Bent Grass
Restionaceae	
Hypolaena fastigiata	Tassle Rope Rush
Leptocarpus tenax	Slender Twine Rush
Xanthorrhoeaceae	
Xanthorrhoea australis	Austral Grass Tree
Pteridophyta	
Dennstaedtiaceae	
Pteridium esculentum	Austral Bracken
Lindsaeaceae	
Lindsaea linearis	Screw Fern
Selaginellaceae	
Selaginella uliginosa	Swamp Selaginella

APPENDIX 2. SPECIES LIST FOR PETER MURRELL NATURE RESERVE

* denotes exotic species

SCIENTIFIC NAME	COMMON NAME
Dicotyledonae	
Asteraceae	
Helichrysum scorpioides	Curling Everlasting
**Hypochoeris radicata*	Cat's Ear
Senecio quadridentatus	
Campanulaceae	
Wahlenbergia spp.	Bluebell
Casuarinaceae	
Allocasuarina monilifera	Necklace She-oak
Dilleniaceae	
Hibbertia acicularis	Prickly Guinea Flower
Hibbertia procumbens	Spreading Guinea Flower
Droseraceae	
Drosera spp.	Drosera species
Epacridaceae	
Epacris impressa	Common Heath
Leucopogon collinus	White Beard Heath
Leucopogon ericoides	Pink Beard Heath
Leucopogon virgatus	Twiggy Beard Heath
Styphelia adscendens	Golden Heath
Euphorbiaceae	
Amperea xiphoclada	Weaver's Plant, Broom Spurge
Fabaceae	
Aotus ericoides	Golden Pea
Bossiaea cinerea	Showy Bossiaea
Dillwynia glaberrima	Smooth Parrot Pea
Gomphalobium huegelii	Bladder Pea
Goodeniacae	
Goodenia lanata	Native Primrose
Haloragaceae	
Gonocarpus tetragynus	Common Raspwort
Lauraceae	
Cassytha glabella	Slender or Tangled Dodder Laurel
Cassytha pubescens	Hairy Dodder Laurel
Myrtaceae	
Baeckea ramosissima	Rosy Heath Myrtle, Baeckea
Eucalyptus amygdalina	Black Peppermint
Leptospermum glaucescens	Glaucous or Soft-fruited Tea-tree
Leptospermum scoparium	Common Tea-tree, Manuka
Pittosporaceae	
Rhytidosporum procumbens	Rhytidosporum

SCIENTIFIC NAME	COMMON NAME
Proteaceae	
Banksia marginata	Honeysuckle, Silver Banksia
Persoonia juniperina	Prickly Geebung
Rubiaceae	
Opercularia varia	Variable Stinkweed
Santalaceae	
Exocarpus cupressiformis	Native Cherry
Stylidiaceae	
Stylidium graminifolium	Trigger Plant
Thymelaeaceae	
Pimelea linifolia	Slender Rice Flower
Tremandraceae	
Tetratheca labillardierei	Leggy Black-eyed Susan
Monocotyledonae	
Centrolepidaceae	
Centrolepis spp.	Centrolepis species
Cyperaceae	
Baumea acuta	Pale Twig Rush
Lepidosperma concavum	Sand or Hill Sword Sedge
Lepidosperma filiforme	Common Rapier Sedge, Thread Rapier Sedge
Schoenus lepidosperma	Slender Bog Rush
Iridaceae	
Patersonia fragilis	Short Purple Flag Iris
Liliaceae	
Laxmannia orientalis	Dwarf Wire Lily
Thelionema caespitosum	Blue Grass Lily
Orchidaceae	
Acianthus spp.	Mosquito Orchid species
Caladenia spp.	Caladenia or Spider Orchid species
Chiloglottis spp.	Bird Orchid species
Eriochilus cucullatus	Parson's Bands
Glossodia major	Waxlip Orchid
Pterostylis spp.	Greenhood species
Pyrorchis nigricans	Fire Orchid or Red Beaks
Thelymitra spp.	Sun Orchid species
Poaceae	
Agrostis spp.	Bent Grass
Austrodanthonia spp.	Wallaby Grass species
Austrostipa spp.	Spear Grass species
**Dactylis glomerata*	Cocksfoot
Deyeuxia quadraseta	
Ehrharta distichophylla	Hairy Rice Grass
**Holcus lanatus*	Yorkshire Fog
**Lolium perenne*	Ryegrass

SCIENTIFIC NAME	COMMON NAME
Poa spp.	Tussock Grass species
Restionaceae	
Acion monocephalum	Cord Rush
Hypolaena fastigata	Tassel Rope Rush
Leptocarpus tenax	Slender Twine-rush
Pteridophyta	
Dennstaedtiaceae	
Pteridium esculentum	Austral Bracken
Schizaeaceae	
Schizaea spp.	Comb Fern species
Selaginellaceae	
Selaginella uliginosa	Swamp Selaginella

APPENDIX 3. SPECIES LIST FOR MT WILLIAM NATIONAL PARK

* denotes exotic species

SCIENTIFIC NAME	COMMON NAME
Dicotyledonae	
Apiaceae	
Xanthosia dissecta	Cut Leaf Xanthosia
Xanthosia pilosa	Wooly Xanthosia
Xanthosia pusilla	Small Xanthosia
Xanthosia tridentata	Hill Xanthosia
Asteraceae	
Argentipallium dealbatum	Everlasting
**Cirsium* spp.	Thistle species
**Cirsium vulgare*	Spear Thistle, Scotch Thistle
Euchiton spp.	Native daisy species
Helichrysum scorpioides	Curling Everlasting
**Hypochoeris radicata*	Cat's Ear
Lagenifera stipitata	Blue Bottle Daisy
Senecio spp.	
Casuarinaceae	
Allocasuarina monilifera	Necklace She-oak
Dilleniaceae	
Hibbertia acicularis	Prickly Guinea Flower
Hibbertia procumbens	Spreading Guinea Flower
Hibbertia prostrata	Prostrate Guinea Flower
Hibbertia riparia	Erect Guinea Flower
Hibbertia sericea	Silky Guinea Flower
Droseraceae	
Drosera peltata ssp *auriculata*	Tall Sundew
Drosera macrantha	Climbing Sundew
Drosera pygmaea	Dwarf Sundew
Epacridaceae	
Brachyloma ciliatum	Ciliate Brachyloma
Epacris impressa	Common Heath
Epacris lanuginosa	Swamp Heath
Epacris obtusifolia	Blunt Leaf Heath
Leucopogon collinus	White Beard Heath
Leucopogon ericoides	Pink Beard Heath
Leucopogon virgatus	Common Beard Heath
Monotoca elliptica	Tree Broom Heath
Monotoca scoparia	Prickly Broom Heath
Sprengelia incarnata	Pink Swamp Heath
Styphelia adscendens	Golden Heath
Euphorbiaceae	
Amperea xiphoclada	Weaver's Plant, Broom Spurge

SCIENTIFIC NAME	COMMON NAME
Fabaceae	
Aotus ericoides	Golden Pea
Bossiaea cinerea	Showy Bossiaea
Dillwynia glaberrima	Smooth Parrot Pea
Gompholobium huegelii	Bladder Pea
Platylobium triangulare	Ivy Flat Pea
Sphaerolobioum minus	Globe Pea
Goodeniaceae	
Dampiera stricta	Blue Dampiera
Goodenia lanata	Native Primrose
Selliera radicans	Swamp Weed
Haloragaceae	
Gonocarpus humilis	Raspwort
Gonocarpus micranthus	Creeping Raspwort
Gonocarpus tetragynus	Common Raspwort
Lauraceae	
Cassytha glabella	Dodder Laurel
Loganiaceae	
Mitrasacme pilosa	Hairy Mitrewort
Mimosaceae	
Acacia genistifolia	Spreading Wattle
Acacia suaveolens	Sweet Wattle
Acacia terminalis	Sunshine Wattle
Myrtaceae	
Baeckea ramosissima	Rosy Heath Myrtle, Baeckea
Calytrix tetragona	Common Fringe Myrtle
Eucalyptus amygdalina	Black Peppermint
Kunzea ambigua	Tick Bush
Leptospermum lanigerum	Woolly Tea-tree
Leptospermum scoparium	Tea-tree, Manuka
Melaleuca squarrosa	Scented Paperbark
Pittosporaceae	
Rhytidosporum procumbens	Rhytidosporum
Polygalaceae	
Comesperma calymega	Blue Spike Milkwort
Proteaceae	
Banksia marginata	Honeysuckle, Silver Banksia
Hakea teretifolia	Dagger Hakea
Rubiaceae	
Opercularia varia	Variable Stinkweed
Rutaceae	
Boronia parviflora	Swamp Boronia
Correa reflexa	Native Fuschia
Sapindaceae	
Dodonaea viscosa	Hop Bush

SCIENTIFIC NAME	COMMON NAME
Stylidiaceae	
Styldium graminifolium	Trigger Plant
Thymelaeaceae	
Pimelea humilis	Common Rice Flower
Pimelea linifolia	Slender Rice Flower
Tremandraceae	
Tetratheca pilosa	Common Lilac Bells
Monocotyledonae	
Cyperaceae	
Baumea acuta	Twigg Rush
Gymnoschoenus sphaerocephalus	Buttongrass
Lepidosperma concavum	Sand or Hill Sword Sedge
Lepidosperma filiforme	Common Rapier Sedge, Thread Rapier Sedge
Lepidosperma gunnii	Sword Sedge
Lepidosperma longitudinale	Pithy or Common Sword Sedge
Schoenus lepidosperma	Slender Bog Rush
Schoenus turbinatus	Bog Rush
Iridaceae	
Patersonia fragilis	Short Purple Flag Iris
Liliaceae	
Burchardia umbellata	Milkmaids
Laxmannia orientalis	
Orchidaceae	
Acianthus spp.	Mosquito Orchid species
Caladenia spp.	Caladenia or Spider Orchid species
Glossodia major	Waxlip Orchid
Pyrorchis nigricans	Fire Orchid or Red Beaks
Pterostylis spp.	Greenhood species
Thelymitra spp.	Sun Orchid species
Poaceae	
**Aira caryophyllea*	Silvery Hair Grass
Austrodanthonia spp.	Wallaby Grass
Austrostipa rudis	Spear Grass
Austrostipa spp.	Spear Grass species
Deyeuxia quadriseta	Bent Grass
Dichelachne spp	Plume Grass species
Ehrharta acuminata	Rice Grass
Ehrharta distichophylla	Hairy Rice Grass
Ehrharta spp	Rice Grass species
Pentapogon quadrifidus	Five-awned Spear Grass
Poa spp.	Tussock Grass species
Restionaceae	
Empodisma minus	Spreading Rope Rush
Eurychorda complanata	Flat Cord Rush

SCIENTIFIC NAME	COMMON NAME
Hypolaena fastigiata	Tassle Rope Rush
Leptocarpus tenax	Slender Twine Rush
Xanthorrhoeaceae	
Xanthorrhoea australis	Austral Grass Tree
Xyridaceae	
Xyris gracilis	Slender Yellow Eye
Pteridophyta	
Schizaeaceae	
Schizaea asperula	Rough Comb Fern
Schizaea spp.	Comb Fern species
Selaginellaceae	
Selaginella uliginosa	Swamp Selaginella
Lindsaeaceae	
Lindsaea linearis	Screw Fern

APPENDIX 4. SPECIES LIST FOR HEATH–PASTURE BOUNDARIES SURVEY

* denotes exotic species, # denotes rare or threatened species

SCIENTIFIC NAME	COMMON NAME
Dicotyledonae	
Aizoaceae	
Carpobrotus rossii	Native Pigface
Apiaceae	
Centella cordifolia	
Eryngium vesiculosum	Prickfoot
Hydrocotyle hirta	Pennywort
Hydrocotyle muscosa	Mossy Pennywort
Xanthosia dissecta	Cut Leaf Xanthosia
Xanthosia pilosa	Wooly Xanthosia
Xanthosia pusilla	Small Xanthosia
Asteraceae	
Ammobium calyceroides	
Argentipallium dealbatum	Everlasting
**Bellis perennis*	Common Daisy
**Chrysanthemoides monilifera*	Boneseed
**Cirsium* spp.	Thistle species
**Cirsium vulgare*	Spear Thistle, Scotch Thistle
Euchiton spp.	Everlasting species
Euchiton collinum	Everlasting
Helichrysum scorpioides	Curling Everlasting
**Hypochoeris radicata*	Cat's Ear
Lagenifera stipitata	Blue Bottle Daisy
**Leontodon taraxacoides*	Hawkbit
Leptinella reptans	
Leptorhynchos squamatus	Scaley Buttons
Ozothamnus scutellifolius	Scale Leaf Bush Everlasting
Senecio glomeratus	Groundsel
Senecio minimus	
Solenogyne spp.	
**Sonchus asper*	Prickly Sow Thistle
**Sonchus oleraceus*	Sow Thistle
**Taraxacum officinale*	Dandelion
**Vellereophyton dealbatum*	
Brassicaceae	
**Raphanus raphanistrum*	Wild Radish
Campanulaceae	
Lobelia alata	Angled Lobelia
Wahlenbergia spp.	Bluebell

SCIENTIFIC NAME	COMMON NAME
Caryophyllaceae	
**Cerastium glomeratum*	Sticky Mouse Ear Chickweed
**Sagina* spp.	
**Stelleria media*	Chickweed
Casuarinaceae	
Allocasuarina littoralis	Black She-oak
Allocasuarina monilifera	Necklace She-oak
Allocasuarina paludosa	Dwarf She-oak
Clusiaceae	
Hypericum gramineum	Small St John's Wort
Hypericum spp.	St John's Wort species
Convolvulaceae	
Dichondra repens	Kidney Weed
Cunoniaceae	
Bauera rubioides	Wiry Bauera
Dilleniaceae	
Hibbertia acicularis	Prickly Guinea Flower
Hibbertia hirticalyx	Guinea Flower
Hibbertia procumbens	Spreading Guinea Flower
Hibbertia prostrata	Prostrate Guinea Flower
Hibbertia riparia	Erect Guinea Flower
Hibbertia sericea	Silky Guinea Flower
Droseraceae	
Drosera peltata ssp *auriculata*	Tall Sundew
Drosera macrantha	Climbing Sundew
Drosera pygmaea	Dwarf Sundew
Drosera spp.	Sundew species
Epacridaceae	
#Acrotriche cordata	Coast Ground Berry
Acrotriche serrulata	Ant's Delight
Astroloma humifusum	Native Cranberry
Astroloma pinifolium	Pine Heath
Brachyloma ciliatum	Ciliate Brachyloma
Epacris impressa	Common Heath
Epacris lanuginosa	Swamp Heath
Epacris obtusifolia	Blunt Leaf Heath
Epacris spp.	Heath species
Leucopogon australis	Spike Beard Heath
Leucopogon collinus	White Beard Heath
Leucopogon ericoides	Pink Beard Heath
Leucopogon parviflorus	Currant Bush, Coast Beard Heath
Leucopogon virgatus	Common Beard Heath
Lissanthe strigosa	Peach Heath
Monotoca elliptica	Tree Broom Heath
Monotoca scoparia	Prickly Broom Heath

SCIENTIFIC NAME	COMMON NAME
Sprengelia incarnata	Pink Swamp Heath
Styphelia adscendens	Golden Heath
Ericaceae	
**Erica lusitanica*	Spanish Heath
Euphorbiaceae	
Amperea xiphoclada	Weaver's Plant, Broom Spurge
Poranthera microphylla	
Ricinocarpus pinifolius	Wedding Bush
Fabaceae	
Almaleea subumbellata	Wiry Bush Pea
Aotus ericoides	Golden Pea
Bossiaea cinerea	Showy Bossiaea
Bossiaea prostrata	Creeping Bossiaea
Daviesia ulicifolia	Native Gorse
Dillwynia cinerascens	Parrot Pea
Dillwynia glaberrima	Smooth Parrot Pea
Dillwynia sericea	Showy Parrot Pea
**Genista* spp.	Broom species
Gompholobium huegelii	Bladder Pea
Hovea linearis	Slender Hovea
Kennedia prostrata	Running Postman
**Lotus* spp.	Trefoil
**Melilotus indicus*	Melilotus
Phyllota diffusa	Tasman Phyllota
Platylobium formosum	Handsome Flat Pea
Platylobium triangulare	Ivy Flat Pea
**Psoralia pinnata*	Blaukeur
Pultenaea daphnoides	Large-leaved Bush Pea
Pultenaea dentata	Clustered Bush Pea, Button Pea
Pultanaea gunnii	Bush Pea
Pultenaea juniperina	Prickly Beauty
Pultenaea paleacea	Bush Pea
Pultenaea stricta	Rigid Bush Pea
Sphaerolobium minus	Globe Pea
**Trifolium repens*	White Clover
**Trifolium* spp.	Clover species
Fumariaceae	
**Fumaria muralis*	Fumitory
Gentianaceae	
**Centaurium erythraea*	Centaury
Geraniaceae	
**Erodium moschatum*	Musk Storksbill
Geranium spp.	Geranium species
Goodeniacae	
Goodenia lanata	Native Primrose

SCIENTIFIC NAME	COMMON NAME
Scaevola hookeri	Creeping Fan Floer
Scaevola spp.	Fan Flower species
Selliera radicans	Swamp Weed
Haloragacaeae	
Gonocarpus humilis	Raspwort
Gonocarpus micranthus	Creeping Raspwort
Gonocarpus tetragynus	Common Raspwort
Lamiaceae	
Mentha diemenica	
Lauraceae	
Cassytha glabella	Dodder Laurel
Cassytha pubescens	Hairy Dodder Laurel
Lentibulariaceae	
Utricularia dichotoma	Fairies' Aprons
Loganiaceae	
Mitrasacme pilosa	Hairy Mitrewort
Mimosaceae	
Acacia baileyana	Cootamundra Wattle
Acacia genistifolia	Spreading Wattle
Acacia gunnii	Ploughshare Wattle
Acacia melanoxylon	Blackwood
Acacia mucronata	Variable Sallow Wattle
Acacia myrtifolia	Myrtle Wattle
Acacia sophorae	Coast Wattle, False Boobyalla
Acacia suaveolens	Sweet Wattle
Acacia terminalis	Sunshine Wattle
#*Acacia ulicifolia*	Juniper Wattle
Acacia verticillata	Prickly Mimosa
Myrtaceae	
Baeckea ramosissima	Rosy Heath Myrtle, Baeckea
Callistemon pallidus	Lemon Bottle Brush
Calytrix tetragona	Common Fringe Myrtle
Eucalyptus amygdalina	Black Peppermint
Eucalyptus nitida	Smithton Peppermint
Eucalyptus ovata	Swamp, Black or Marrawah Gum
Kunzea ambigua	Tick Bush
Leptospermum glaucescens	Glaucous or Soft-fruited Tea-tree
Leptospermum laevigatum	Coastal Tea-tree
Leptospermum lanigerum	Woolly Tea-tree
Leptospermum scoparium	Tea-tree, Manuka
Melaleuca ericifolia	Swamp Paperbark
Melaleuca gibbosa	Bracelet Honey Myrtle
Melaleuca squamea	Purple Honey Myrtle
Melaleuca squarrosa	Scented Paperbark

SCIENTIFIC NAME	COMMON NAME
Onagraceae	
Epilobium spp.	Willowherb
Oxalidaceae	
Oxalis perennans	Wood Sorrell
Pittosporaceae	
Billardiera scandens	Apple Berry
Bursaria spinosa	Prickly Box
Rhytidosporum procumbens	Rhytidosporum
Plantaginaceae	
**Plantago coronopus*	Buck's Horn Plantain
**Plantago lanceolata*	Narrow Leaf Plantain
**Plantago* spp.	Plantain species
Polygalaceae	
Comesperma calymega	Blue Spike Milkwort
Comesperma retusum	Purple Milkwort
Comesperma volubile	Blue Love Creeper
Polygonaceae	
**Acetosella vulgaris*	Sheep's Sorrell
Primulaceae	
**Anagallis arvensis*	Scarlet Pimpernel
Proteaceae	
Banksia marginata	Honeysuckle, Silver Banksia
#*Banksia serrata*	Saw Leaf Banksia
Hakea spp.	Hakea species
Hakea teretifolia	Dagger Hakea
#*Isopogon ceratophyllus*	Horny Cone Bush
Lomatia tinctoria	Guitar Plant
Persoonia juniperina	Prickly Geebung
Ranunculaceae	
Ranunculus spp.	Buttercup species
Rosaceae	
Acaena novae-zelandiae	Buzzy
**Prunus armeniaca*	Apricot
**Rosa rubiginosa*	Rosehips
**Rubus* spp. agg.	Blackberry
Rubiaceae	
Opercularia varia	Variable Stinkweed
Rutaceae	
Boronia parviflora	Swamp Boronia
Boronia pilosa	Fine Leaf Boronia
Philotheca virgata	Twiggy Wax Flower
Santalaceae	
Exocarpos cupressiformis	Native Cherry
Exocarpos strictus	Dwarf Cherry

SCIENTIFIC NAME	COMMON NAME
Sapindaceae	
Dodonaea viscosa	Hop Bush
Scrophulariaceae	
**Parentucellia viscosa*	
Veronica spp.	Speedwell species
Stackhousiaceae	
Stackhousia spp.	
Stylidiaceae	
Stylidium graminifolium	Trigger Plant
Thymelaeaceae	
#*Pimelea flava*	Rice Flower
Pimelea humilis	Common Rice Flower
Pimelea linifolia	Slender Rice Flower
Pimelea spp.	Rice Flower species
Tremandraceae	
Tetratheca labillardierei	Leggy Black-eyed Susan
Tetratheca pilosa	Common Lilac Bells
Violaceae	
Viola hederacea	Ivy Leaf Violet
Monocotyledonae	
Centrolepidaceae	
Centrolepis strigosa	Centrolepis
Cyperaceae	
Baumea acuta	Pale Twig Rush
Baumea juncea	Bare Twig Rush
Baumea tetragona	Twig Rush
Carex inversa	
Gahnia grandis	Cutting Grass
Gahnia sieberiana	Red Fruit Saw Sedge
Gahnia spp.	Cutting Grass species
Gahnia trifida	
Gymnoschoenus sphaerocephalus	Buttongrass
Isolepis fluitans	Floating Club Rush
Isolepis inundata	Club Rush
Isolepis nodosa	Knobby Club Rush
Lepidosperma concavum	Sand Sword Sedge
Lepidosperma elatius	
Lepidosperma ensiforme	
Lepidosperma filiforme	Common Rapier Sedge, Thread Rapier Sedge
Lepidosperma longitudinale	Pithy Sword Sedge
Schoenus apogon	Common Bog Rush
Schoenus lepidosperma	Slender Bog Rush
Schoenus maschalinus	Bog Rush
Schoenus spp.	Bog Rush species

SCIENTIFIC NAME	COMMON NAME
Schoenus tesquorum	Bog Rush
Tetraria capillaris	
Iridaceae	
**Crocosmia Xcrocosmiiflora*	Montbretia
Patersonia fragilis	Short Purple Flag Iris
Patersonia occidentalis	Long Purple Flag Iris
**Romulea rosea*	Onion Grass
Juncaceae	
**Juncus articulatus*	
Juncus australis	
Juncus bufonius	Toad Rush
Juncus krausii	Sea Rush
Juncus pallidus	Pale Rush
Juncus pauciflorus	Rush
Juncus planifolius	Rush
Juncus spp.	Rush species
Luzula flaccida	Woodrush
Luzula spp.	Woodrush species
Liliaceae	
Arthropodium milleflorum	Vanilla Lily
Burchardia umbellata	Milkmaids
Dianella revoluta	Black-anther Flax Lily
#*Dichopogon strictus*	Chocolate Lily
Laxmannia orientalis	Dwarf Wire Lily
Orchidaceae	
Acianthus spp.	Mosquito Orchid species
Caladenia spp.	Caladenia or Spider Orchid species
Chiloglottis spp.	Bird Orchid species
Corybas spp.	Helmet Orchid species
Cryptostylis spp.	Tongue Orchid species
Glossodia major	Waxlip Orchid
Pterostylis spp.	Greenhood species
Pyrorchis nigricans	Fire Orchid or Red Beaks
Thelymitra spp.	Sun Orchid species
Poaceae	
Agrostis aemula	Blown Grass
#*Agrostis australiensis*	Southern Bent
Agrostis avenacea	Bent
**Agrostis capillaris*	Browntop Bent
Agrostis spp.	Bent Grass species
**Aira caryophyllea*	Silvery Hair Grass
**Aira praecox*	Hair Grass
**Anthoxanthum odoratum*	Sweet Vernal Grass
Austrodanthonia caespitosa	Common Wallaby Grass
Austrodanthonia penicillata	Wallaby Grass

SCIENTIFIC NAME	COMMON NAME
Austrodanthonia pilosa	Velvet Wallaby Grass
Austrodanthonia racemosa	Wallaby Grass
Austrodanthonia setacea	Bristly Wallaby Grass
Austrostipa flavescens	Spear Grass
Austrostipa mollis	Spear Grass
Austrostipa pubinodis	Spear Grass
Austrostipa rudis ssp *australis*	Spear Grass
Austrostipa spp.	Spear Grass species
**Briza minor*	Quaking Grass
**Bromus willdenowii*	Prairie Grass
**Bromus hordeaceus*	Brome
**Cynodon dactylon*	Couch Grass, Bermuda Grass
**Cynosurus echinatus*	Dogstail
**Dactylis glomerata*	Cocksfoot
Deyeuxia quadriseta	Bent Grass
Dichelachne rara	Scarce Plume Grass
Dichelachne spp	Plume Grass species
Ehrharta acuminata	Rice Grass
Ehrharta distichophylla	Hairy Rice Grass
Ehrharta spp	Rice Grass species
Ehrharta stipoides	Weeping Grass
**Festuca arundinacea*	Tall Fescue
Hemarthria uncinata	
**Holcus lanatus*	Yorkshire Fog
**Lolium perenne*	Ryegrass
**Lolium rigidum*	Stiff Ryegrass
Notodanthonia semiannularis	Wallaby Grass
Pentapogon quadrifidus	Five-awned Spear Grass
**Poa annua*	Winter Grass
Poa labillardieri	Tussock Grass
Poa poiformis	Blue Tussock Grass
**Poa pratensis*	Kentucky Bluegrass
Poa sieberiana	Tiny Tuft
Poa spp.	Tussock Grass species
Poa tenera	Tussock Grass
**Stenotaphrum secundatum*	
Themeda triandra	Kangaroo Grass
**Vulpia bromoides*	Squirrel Tail Fescue, Hair Grass, Silk Grass
Zoysia macrantha	
Restionaceae	
Acion hookeri	Cord Rush
Acion monocephalum	Cord Rush
Calorophus elongatus	Rope Rush
Empodisma minus	Spreading Rope Rush
Eurychorda complanata	Flat Cord Rush

SCIENTIFIC NAME	COMMON NAME
Hypolaena fastigiata	Tassel Rope Rush
Leptocarpus tenax	Slender Twine Rush
Lepyrodia muelleri	
Xanthorrhoeaceae	
Lomandra longifolia	Sagg
Xanthorrhoea australis	Austral Grasstree
Xyridaceae	
Xyris gracilis	Slender Yellow Eye
Xyris marginata	Emarginate Yellow Eye
Pteridophyta	
Dennstaedtiaceae	
Pteridium esculentum	Austral Bracken
Gleicheniaceae	
Gleichenia microphylla	Scrambling Coral Fern
Lindsaeaceae	
Lindsaea linearis	Screw Fern
Lycopdiaceae	
Lycopodiella serpentina	Bog Clubmoss
Schizaeaceae	
Schizaea spp.	Comb Fern species
Selaginaceae	
Selaginella uliginosa	Swamp selaginella